Factors Influencing Health of Ecosystems

Key Terms Used in This Section

Disturbance — Refers to events that alter the structure, composition, or function of terrestrial or aquatic habitats. Natural disturbances include, among others, drought, floods, wind, fires, wildlife grazing, and insects and diseases. Human-caused disturbances include, among others, actions such as timber harvest, livestock grazing, roads, and the introduction of exotic species.

Excessive livestock grazing pressure — Grazing pressure that results in a decline in physiological vigor of plants, typically observed as a decline in reproductive output (for example, seeds and rhizomes) and growth, both above ground (for example, tiller production of grasses) and below ground (for example, root growth). This decline in physiological vigor results in decreased ability of the plant to compete for resources and results in alteration of plant species composition in plant communities. The connotation of this phrase is negative.

Exotic species — A plant or animal species introduced from a distant place; not native to the area.

Habitat Fragmentation — The break-up of a large land area (such as forest) into smaller patches isolated by areas converted to a different land type. The opposite of connectivity.

Shade-intolerant — Species of plants that do not grow well in or die from the effects of too much shade. Generally these are fire-tolerant species.

Succession — A predictable process of changes in structure and composition of plant and animal communities over time. Conditions of the prior plant community or successional stage create conditions that are favorable for the establishment of the next stage. The different stages in succession are often referred to as seral stages.

Introduction

The ecosystem conditions presented in earlier sections of this chapter have been influenced or caused by a variety of interrelated factors such as fire suppression, timber harvest, human demographics, insects and disease, roads, livestock grazing, and noxious weeds. Many of these factors influence more than one resource or vegetation type—that is, they create predictable conditions that can affect a number of ecosystem resources regardless of whether the vegetation is forestland, rangeland, aquatic, or riparian area. They also affect each other, and their effects often cannot be separated.

For example, livestock grazing influences the dry forest, riparian, cool shrub, dry shrub, and dry grass potential vegetation groups, but it also influences moist and cold forests for a much shorter time period during the course of a year. Livestock grazing also affects the fire regimes in forests as well as rangelands through disruption of fine surface fuels. Fire regimes in turn influence livestock grazing through effects on vegetation. Roads can influence aquatic and riparian conditions, terrestrial wildlife habitats, and the spread of noxious weeds, which in turn can affect conditions in rangelands, forestlands, and wildlife habitats. Most of the factors operate across subbasins and are landscape-based.

These factors have been discussed as appropriate in individual sections of this chapter with regard to their direct or indirect influence on separate components of the ecosystems of the interior Columbia Basin. This section presents a more integrated discussion of the influence of various factors on ecosystem health in the project area.

Fire and Fire Suppression

Historical to Current Trends

Wildfire has long been a dominant disturbance in the interior Columbia River Basin, affecting succession in the native system. American Indians' use of fire as a disturbance process had an integral role on the landscape over vast areas of the basin for at least the past 2,000 years (Mehringer et al. 1977, Ross 1981, Shinn 1980, and Woods and Horstman 1996). The use of fire substantially augmented the extent and incidence of wildfires, and the effects on the landscape were especially noticeable near grasslands, low-elevation forests, and in or near major valleys or other significant settlement locations, resource acquisition areas, and travel routes (Barrett and Arno 1982). Elsewhere, lightning supplied the ignition source for wildfires that burned frequently in dry forests and moist rangelands and less frequently in moist forests and drier rangelands. Some of the highly variable factors that influenced the natural fire regime in forests and rangelands were extent and water content of fuel, topography, and weather.

Reduced fire occurrence began in the late 1800s as a result of the following: (1) relocation of American Indians; (2) fuel removal by excessive livestock grazing; (3) disruption of fuel continuity on the landscape due to irrigation, cultivation, roads, and community development; and (4) adoption of a fire exclusion policy.

Early in the 20th century, wildfires began to be perceived as dangerous, destructive, and undesirable. Early wildfire suppression efforts were crude but somewhat successful in low to mid elevations because of low levels of fuels, which had been maintained by the predominant fire regimes.

Wildfire suppression activities, aided by improved technology for fire detection, prevention, and suppression, were generally successful in reducing the extent of wildfires from the 1910s through the 1960s. Fuel loadings have steadily increased as a result of suppression efforts and fire frequencies have declined (Agee 1993). As a result, fire size, intensity, and severity have increased, as have suppression costs and the associated hazards to life and property.

The area burned by wildfires in the basin steadily increased between the 1970s and 1990s, even though land managers have been allocating increasing amounts of resources to wildfire suppression. The current extent of wildfires is approaching that experienced in the early 1900s. The average costs of wildfire suppression, number of firefighter fatalities, and extent of high-intensity fires during the past 25 years are double the corresponding levels that occurred between 1910 and 1970. Further complicating matters, human populations within the urban–rural–wildland interface have substantially increased within the past few decades. These areas of rapidly-growing human populations are commonly associated with high fire risks. (See the Urban–Rural–Wildland discussion, later in this section, for additional details.)

Wildfire suppression, in conjunction with other factors, has caused great changes in: disturbance frequency, size, and severity; vegetation structure, density and composition; and the resulting patches and patterns. These are different from those to which native plant and animal species have adapted. Fire exclusion has caused a shift to conditions with more severe disturbance regimes. Current amounts of fire in these areas are generally less than in the native system, but when current wildfires occur they are much larger in patch size and more severe in their effects compared to the native system. In addition the resistance to control is substantially higher than conditions of the early 1900s. Hann , Jones, Karl, et al. (1997) called this increase in wildfire size, severity and resistance to control, “uncharacteristic wildfire effects” (Map VB14.1).

In the past 100 years, fires have become less frequent and more intense, except in dry grass and dry shrub PVGs that have been invaded by exotic annual grasses, where fire disturbance has become more frequent.

Overview of Fire Suppression Influence

There is little similarity between the historical and current succession / disturbance regimes within forest and rangeland systems. In the past 100 years, fires have become less frequent and more intense (Agee 1993, Gast et al. 1991 in Lehmkuhl et al. 1994). Exceptions to this general trend are the dry grass and dry shrub PVGs that have been invaded by exotic annual grasses. In these instances, fire disturbance has become more frequent.

In forested potential vegetation groups (PVGs), wildfire suppression coupled with timber harvest, introduced pathogens, livestock grazing, and natural succession are responsible for these changes over the past 100 years (Hann, Jones, Karl, et al. 1997). The most notable changes in *forestlands* include:

1. Declines in extent and increasing fragmentation of dry and moist late seral forests, especially single storied;
2. Declines in the extent of early seral forests;
3. Dramatic increases in the extent and connectivity of mid seral forests;
4. Large declines in the shade-intolerant cover types such as ponderosa pine, western larch, western white pine, and whitebark pine, which have become more fragmented; and increases of shade-tolerant forests such as Douglas-fir, grand fir, white fir, and subalpine fir, which have become significantly less fragmented;
5. Overall forest composition and structures largely becoming more homogeneous; and
6. Decline in the number of large trees and snags in harvested and roaded areas (Hann, Jones, Karl, et al. 1997).

In forested PVGs, fire severity has shifted substantially from nonlethal to lethal between the historical and recent past on Forest Service- and BLM-administered lands (Hann, Jones, Karl, et al. 1997). Lack of frequent nonlethal underburns has resulted in increases in fuel loading, an increase in duff depth (up to 24 inches under old trees), an increase in stand density (generally development of dense conifer understories beneath old stands and thickets of small trees where the overstory has been removed), an increase in shade-tolerant species, and fuel ladders that can carry fire from the surface into the tree crowns. In general, the exclusion of fire and extensive harvesting of large, shade-intolerant trees resulted in a shift of forest dominance to smaller, shade-tolerant trees that were more susceptible to wildfire, stress, insects, and diseases (Hann, Jones, Karl, et al. 1997).

Areas dominated by rangeland PVGs also had substantial changes in disturbance regime patterns as a result of wildfire suppression and other factors such as agriculture, excessive livestock grazing pressure, and the introduction of exotic plants. The most notable changes on *rangelands* are:

1. Shifts from all rangeland PVGs, especially dry grass and dry shrub PVGs, to agricultural PVG;
2. Encroachment of woody species such as sagebrush, juniper, ponderosa pine, and Douglas-fir, especially in the dry grass and cool shrub PVGs, which has reduced herbaceous understory and biodiversity;
3. Increased densities of sagebrush in the dry shrub PVG, leading to a decline in the herb and grass understories;
4. In some places, replacement of native cover types by exotic species, increasing soil erosion, simplifying stand structure, reducing biodiversity, and reducing utility to wildlife species; in many locations, the altered fire regime continues the dominance of exotic annual grasses; and
5. Increased fragmentation and loss of connectivity within and between blocks of habitat, especially in the shrub-steppe and riparian areas (Hann, Jones, Karl, et al. 1997).

In rangelands as in forests, these changes in the fire regime have caused greater homogeneity, or simplification, of many landscapes. In dry grasslands where fire typically has been absent, shrubs are more competitive than grasses, in part because shrubs have deeper root systems than grasses, allowing them to tap soil moisture in dry years. This change in disturbance regime caused the shift from herb to shrub or woodland, or from shrub to woodland. When the changes in disturbance regime shifted herblands to exotics, the result was a short-cycle regime, particularly if the exotics were highly-flammable exotic annual grasses. This increased fire frequency has caused a loss of shrub cover, particularly sagebrush and bitterbrush, and reduction in bunchgrasses (Leonard and Karl 1995a, 1995b).

Landscapes that are dominated by a *mosaic of forest and rangeland PVGs* had inherently more diverse disturbance regimes than did forest-dominated or rangeland-dominated landscapes alone. Although the individual landform, potential vegetation group, and succession / disturbance regime relationships and resulting changes to vegetation patterns are generally the same as in the forest-dominated or rangeland-dominated landscapes, the effects at the landscape level are substantially different. The forest–rangeland landscape pattern has many complex ecotonal and disturbance relationships that are further complicated by the spreading influences among its varied environments and communities. In general, the changes that occurred in forest–rangeland landscape patterns have been more substantial than those observed in either the forest or rangeland landscape patterns alone. This is because the energy gradients are steeper, topography is more rugged, the disturbance regimes are more dynamic because of the forest / rangeland mosaic, and the diversity of species is higher with forest–rangeland landscape pattern (Hann, Jones, Karl, et al. 1997).

Specific Influences of Fire Suppresssion

Fire Suppression in the Cold Forest PVG

Cold forests have longer fire intervals than dry or moist forests, so the effects of fire exclusion on forest structure and composition are not as noticeable as in the other forest PVGs. The cold climate and short growing season in cold forests also slow the natural rate of change in vegetation when compared to dry or moist forests. However, some changes from historical conditions have occurred.

Historically, the mixed-fire regime in the cold forest PVG reduced fuels and thinned tree densities, thereby accelerating the growth rate of survivors. Under fire exclusion policies, the fire interval has increased and intermediate nonlethal underburns (which along with stress, insects, and disease, used to thin the forests and accelerate the growth in surviving trees) have been lost. When fires do occur they tend to be larger, lethal, crown-fire events of high intensity (particularly on upland slope environments) because of increased tree densities and fuel loading, both due to wildfire suppression; the changes in the fire regime resulted in changes to landscape structure and composition. Maintenance of dead and downed wood on these sites is important for nutrient cycling; therefore, the severity of wildfires can have long-lasting impacts on soils and site productivity (Hann, Jones, Karl, et al. 1997).

With fire exclusion, more areas of lodgepole pine are in a late seral multi-story structure, a stage more susceptible to outbreaks of mountain pine beetle. This leads to larger areas of mountain pine beetle outbreaks, for longer periods, and often with greater intensity than occurred historically. Increasing size of susceptible stands of trees has also contributed to higher levels of other insects and diseases (Hann, Jones, Karl, et al. 1997).

Historically, shade-intolerant species dominated regeneration and young forest environments. This relationship has been altered, resulting in landscapes that now have mixed dominance or are dominated by shade-tolerant species, such as extensive areas where conifers have replaced or are replacing aspen. This is especially true where fire exclusion have favored the establishment of shade-tolerant species. As a result, many areas are highly susceptible to tree mortality from fire, insects, disease, and stress.

In particular, loss of whitebark pine and alpine larch habitat, due to white pine blister rust and overstocking resulting from fire exclusion, has become a forest health concern in the past ten years (Hann, Jones, Karl, et al. 1997). Fire exclusion has allowed the encroachment of shade-tolerant trees to form dense stands and fuel ladders, and it has precluded the regeneration of whitebark pine seedlings, with immense consequences to cold forest ecosystems. For example, grizzly bears depend on whitebark pine seeds as a major component of their diet because the seeds are large, are a good source of protein, and are available in squirrel caches. What the decline in whitebark pine means to grizzly bears in the long term cannot yet be determined, but grizzly bears are a species of special importance to tribes and are listed under the Endangered Species Act. Other cold forest species of importance to tribes that have been detrimentally affected by fire suppression, increased stand density, decreasing shrubs, and large tree components include: blue grouse, spruce grouse, and snowshoe hare.

Fire Suppression in the Moist Forest PVG

The moist forest potential vegetation group has a productive environment which rapidly produces biomass and accumulates fuels. The effective exclusion of almost all nonlethal underburns and a reduction of mixed fires has resulted in the development of dense multi-storied stands with high potential for stand-replacing fires. These highly productive forests have increased amounts of carbon and nutrients stored in woody material, resulting in fires that are of higher intensity and severity. Even where fires do not crown, dominant trees can be killed by consumption of large diameter surface fuels and duff layers. Potential for high amounts of soil heating and death of tree roots and other understory plants is much higher than it was historically.

The current fire regime in the moist forest has become simplified compared to the historical regime. As a result of higher fuel loads, increased stocking levels of trees, and high late summer water stress levels, most of the moist forest PVG shifted to lethal crown fire or mixed fire regimes. With recent fire suppression efforts, the general fire interval has almost tripled. Increasing fire intervals without corresponding fuel reduction, together with the elimination of the thinning effect that historically reduced shade-tolerant trees in the stands, has resulted in higher-intensity fires.

Fire exclusion has led to a decrease in extent of late seral single and multi-story structures and a decline in early seral forest. Mid seral forest has increased, especially the shade-tolerant cover types. Change of potential insect and pathogen disturbances is directly correlated with the change in composition, structure, and connectivity of forest host species. Thus, increases in insect and pathogen disturbances are strongly tied to the increase in shade-tolerant species, dominance of medium to large trees, increased crown cover, development of the understory, and development of multiple crown layers. Causal factors vary geographically by type and intensity of timber harvest, effectiveness of fire exclusion, and the resultant stand structures and composition.

Similar to changes in dry forest systems of the project area, susceptibility to large-scale damage by insect infestations and diseases has increased in many moist forests. Tree density has increased and vigor has decreased in moist Douglas-fir and grand fir forests, making them more susceptible to insect and disease damage.

Tree harvest and white pine blister rust have all but eliminated the western white pine cover type. In its place are Douglas-fir, grand fir, and white fir. This has had a huge impact on the structure and fire ecology of the moist forest, because no other tree species can grow as fast or as tall as, or fill the ecological niche of, western white pine. This affects the usability of habitat for species such as the pygmy shrew, wolverine, Yuma myotis (bat), long-eared myotis, fringed myotis, and long-legged myotis (Wisdom et al. in press). Economically, western white pine is a highly desirable species.

Soil fertility of some sites has been depleted through timber harvest practices or through multiple fires, which displace or erode surface soil or remove much of the large woody material, litter, or duff. Fire exclusion also reduces site productivity, increases the probability of insect and disease infestation, increases the probability of high intensity fires, and changes habitat conditions (Hann, Jones, Karl, et al. 1997). In general, moist forests identified as having the most forest health problems are in areas that have been roaded and harvested. This is because fire suppression has been most effective in roaded areas and consequently changed the vegetation composition and structure within those areas.

With current trends in moist forests, tribes have experienced associated declines in some of the plants that they consider important, such as huckleberry, buffaloberry, and beargrass.

Fire Suppression in the Dry Forest PVG

Current fire regimes in the dry forest potential vegetation group are the least similar to historical regimes of all the forest PVGs. This is partly because dry forests are more accessible to housing developments, logging, and grazing. Dry forests also contain tree species historically favored by the timber market (Everett et al. 1994), and they were subject to disruption of natural fires through suppression activities.

The interval between fires has doubled, tripled, or more. Increasing the intervals without corresponding fuel reductions has resulted in much higher fuel loads and fire intensities than were previously experienced. With the exclusion of fire, stands are often more dense, which means larger amounts of carbon are tied up in woody materials. Overstocked stands result in moisture stress in the normal summer drought period, and make stands highly susceptible to insects such as bark beetles.

Historically, fires not only favored the regeneration and release of shade-intolerant species by providing openings and bare mineral soil, but they also minimized fuel loads and effectively thinned from below, favoring lower tree densities and drought and disease tolerance. Fires also rejuvenated shrubs in the dry forest. Lack of frequent fire and the resulting dense forests have caused declines in some shrubs that are important to tribes such as mountain mahogany, chokecherry, and serviceberry. Species composition has changed to dominance by trees such as Douglas-fir, grand fir, and white fir. The younger forest structure or multi-storied structure composed of a high proportion of shade-tolerant species is highly susceptible to large-scale infestations of insects and disease (Hann, Jones, Karl, et al. 1997).

Current fire regimes in the dry forest potential vegetation group are the least similar to historical regimes of all the forest PVGs.

Bark beetles currently often replace fire in eliminating trees growing in excess of site potential. Outbreaks of western pine beetle and mountain pine beetle have become more intensive and extensive. Susceptibility to the Douglas-fir beetle has increased in many areas compared to historical conditions. This can be attributed to increased spread of shade-tolerant Douglas-fir, increased abundance of host trees of adequate size for successful bark beetle breeding, increased patch densities and layering of canopies, and increased landscape contiguity of susceptible areas. Susceptibility to fir engraver beetle has increased in many areas because of expansion of grand fir and white fir and expansion of multi-layered understories. Spruce beetle activity appears to be correlated with the drought of the past eight to nine years (Hann, Jones, Karl, et al. 1997).

Increasing susceptibility to Douglas-fir dwarf mistletoe was associated with increased abundance of Douglas-fir, increased canopy layering, and Douglas-fir encroachment on dry and relatively moist sites that historically had frequent understory fires. Increases in susceptibility to root diseases are associated with effective fire exclusion, the selective harvest of shade-intolerant species, and the contagious spread of Douglas-fir and true firs in dense, multi-story arrangements (Hann, Jones, Karl, et al. 1997).

The increasing number of small dead trees in stands attacked by insects and diseases makes forests even more susceptible to large high-intensity fires. The stands that are most susceptible to moisture stress, insects, and disease tend to be those at the lowest elevations, which typically border private, state, tribal, or other land ownerships. The clumpy character of historical stands that was created by fire has changed. Overall, stand structures changed from open park-like stands of large trees with clumps of small trees, to dense overstocked young stands with several canopy layers (Caraher et al. 1992, Gast et al. 1991 in Lehmkuhl et al. 1994). As dry forests become denser, moisture and light become more limiting and openings less common, and tribes have seen declines in plants that they consider important, such as chokecherry, serviceberry, bitterroot, and biscuitroot.

Fire exclusion effects have been greatest in the most heavily roaded areas where suppression has been successful. Development of residential areas and other cultural facilities in project area forests has been most common in this PVG, which, coupled with the changed fire regime, has caused a greatly increased risk to life and property. Homes, private, tribal, and state forest resources, wildlife winter ranges, and other important resources are increasingly at risk from fire and insect and disease attack from lands administered by the BLM and Forest Service (Everett et al. 1994).

Fire suppression has helped to shift habitats away from species that require open stands and to favor those species needing dense stands. Species that require late seral habitats—such as whiteheaded woodpecker, white breasted nuthatch, and western gray squirrel—are finding habitat scarce in much of the basin, while species that can use mid seral structures have an abundance of habitat. However, fire suppression has also resulted in a reduction of early successional stages so these habitats are still in short supply in many areas. Large, intense fires have also created substantial amounts of habitat for snag-dependent species; however, this increase is short-lived and fires may lead to long-term shortages of snags over large areas. The increased intensity of fires can have adverse effects on litter and downed wood, which can have adverse effects on amphibians. Within burned areas, mosaic patterns of habitat and unburned islands of vegetation have decreased, probably limiting the distribution of less mobile species.

Fire Suppression in the Cool Shrub PVG

Current fire regimes in the cool shrub potential vegetation group are the closest to historical of all of the rangeland PVGs. Generally, fire regimes have decreased in frequency and increased in intensity, resulting in a decline in extent of upland herblands and upland shrublands and an increase in extent of upland woodlands. Juniper woodlands have greatly expanded at the expense of both upland herbland and upland shrubland, hastened by excessive livestock grazing pressure. The expansion of western juniper is causing decreases in understory productivity, decreases in diversity, changes in the hydrologic cycle, and habitat conversion. Habitat is changing in favor of such species as ash-throated flycatcher, bushtit, and spotted bat, at the expense of species such as ferruginous hawk, burrowing owl, and lark sparrow.

When woodland encroachment occurred, fuels accumulated and communities exhibited high stress and low foliage moisture. During drought years, very intense fire events had the potential to occur in woodlands, and often caused relatively severe effects on the soil surface and mortality to the understory grasses and forbs. Today, the upland herblands in the cool shrub PVG has been nearly eliminated. Under the current dynamics, the cool shrub PVG is susceptible to invasion by exotic weed species that could eventually dominate at least two percent of the PVG (Hann, Jones, Karl, et al. 1997).

The increase in density and extent of western juniper woodlands has had beneficial effects as well, such as to several wildlife species. Western juniper is important to and used extensively by tribes.

Fire Suppression in the Dry Shrub PVG

Averaged for the entire dry shrub group, the current fire regime has not changed much from the historical regime. However, fire frequency has increased in locations where exotic annual grasses have invaded, and it has decreased elsewhere. Throughout the dry shrub PVG, fire severity has increased since historical times (Hann, Jones, Karl, et al. 1997).

In those areas where fire frequency has *decreased*, trees (in the absence of fire) can invade dry shrub areas that lie adjacent to woodland or dry forest areas. Tree establishment in the dry shrubland PVG disrupts the established nutrient regime. Tree species tie up nitrogen and other trace nutrients, thereby decreasing overall site productivity. Subsequently, foliage cover, basal cover, and litter from shrubs, grasses, and forbs decline, thereby exposing surface soil and increasing erosion potential. Erosion is potentially aggravated by excessive grazing pressure from livestock and big game. Once the surface soil is eroded and the subsoil exposed, the environment is more conducive to tree species that are more competitive for sub-soil moisture (Hann, Jones, Karl, et al. 1997).

When woodland encroachment occurs, fuels accumulate and communities exhibit high stress and low foliage moisture. During drought years, very intense fire events can occur in areas of woody encroachment, often causing relatively severe effects to the soil surface and mortality to the understory grasses and forbs.

In those areas where fire frequency has *increased*, the eventual result is mortality of perennial species and prevention of their recruitment because most perennial vegetation in the dry shrub PVG is not adapted to more frequent, high-intensity fires. Hann, Jones, Karl, et al. (1997) expect an expansion of this shift in fire regimes over the next 50 years. Such an expansion will convert even more of the dry shrub PVG to annual exotic grasses, which can result in altering ecosystem processes (Vitousek et al. 1996) because the additional abundance of fine, flash-type fuels, combined with sagebrush fuels, creates intense fires. Some of the potentially altered processes include primary productivity, decomposition and nutrient cycling, hydrology, and disturbance regimes.

For example, cheatgrass, an annual grass, is well adapted to frequent fire regimes. Standing cheatgrass and litter produced by cheatgrass are extremely flammable, so cheatgrass helps to maintain the frequent fire return intervals (Billings 1948) in places where they did not occur before. Pellant (1996) calls this the "cheatgrass–wildfire cycle." (See discussion of cheatgrass, later in this section, for additional details.)

As a result of the cheatgrass-wildfire cycle, big game winter range has declined, habitat supporting the densest population of nesting raptors in North America has declined, the persistence of native plants is threatened, non-game bird abundance has declined, native species richness has declined, successional recovery periods have been extended, and presence of biological crust organisms has declined (Hann, Jones, Karl, et al. 1997). Declining species important to tribes in the dry shrub PVG are the jackrabbit, pygmy rabbit, sage grouse, and sharptailed grouse.

Fire Suppression in the Dry Grass PVG

Because of fire suppression and excessive livestock grazing pressure, fire intervals in the dry grass potential vegetation group have increased, as have overall fire intensities. Fuels generally did not increase on the upland herblands because they typically were grazed, but the vigor of the dominant grasses and forbs decreased in these types because of the continued absence of fire. Fuels accumulate when shrubland and woodland encroachment occurs. During drought years, very intense fire events can occur in areas of woody encroachment, often causing relatively severe effects on the soil surface and mortality to the understory grasses and forbs. Effects of increased fire intensities include decline in presence of biological crust organisms in the dry grass PVG (Hann, Jones, Karl, et al. 1997).

Fire Suppresssion in Riparian PVGs

Within riparian woodlands, the abundance of mid seral vegetation has increased while the extent of late and early seral structural stages has decreased, primarily because of fire exclusion and harvest of large trees. Within riparian shrublands, there has been extensive spread of western juniper and exotic grasses and forbs. Overall, there has been a decrease in large trees and late seral vegetation in riparian areas (Lee et al. 1997).

This change in habitat has had detrimental effects on the silver-haired bat, the hoary bat, the northern flying squirrel, and many other species. Lack of fire, along with excessive livestock grazing pressure, has been a factor in the decline in willows and cattails, which are important to tribes.

Fire Suppression in Aquatic Areas

Present aquatic systems in the basin have evolved in response to and in concert with wildfire. The effects of fire on aquatic systems may be direct and immediate (for example, increased water temperature, or chemical input) or indirect occurring over an extended period, but ultimately fire results in a natural mosaic of habitats and populations. The intensity and scale of these effects are related to the size and intensity of fire, geology, topography, size of the stream system, and amount, intensity, and timing of subsequent precipitation events (Lee et al. 1997).

Physical properties of soil that influence water retention are altered by heating. In some cases, soils become water repellent after severe burns (McNab et al. 1989). The amount of vegetation remaining in a watershed after a fire directly influences runoff and erosion by physically mediating the force of precipitation on soil surfaces, altering the evapotranspiration cycle, and providing soil stability through root systems. Runoff rate and pattern and subsequent erosion potential are directly affected by the amount of organic debris left in the watershed (Lee et al. 1997). According to Wells et al. (1979 cited in Jensen et al. 1997), intense fire can have four generally negative impacts on soils:

1. Removal of protective surface layer organic materials;
2. Volatilization of large amounts of nitrogen and smaller amounts of other nutrients;
3. Conversion of some nutrients into soluble forms that can be lost by leaching; and
4. Heating of the soil and alteration of its physical, chemical, and biological properties. In general, the hotter the burn the greater the potential for soil damage and nutrient loss (Jensen et al. 1997).

The main effects on aquatic habitats from more intense fire regimes in the uplands and riparian areas compared to historical times come from more thorough consumption of ground cover that would protect the soil against rain, wind, and overland flow; more thorough killing of the overstory and understory plants that would stabilize the soil; and intense fire conditions over more of the landscape. The result is more sedimentation and degradation of aquatic habitats than the aquatic species evolved with, spread over a larger area. There is an increased risk of mass sedimentation events until watershed soils have time to stabilize and vegetation has time to recover following a wildfire. In cases where plant and animal populations are severely affected, there is less opportunity for surrounding populations to re-invade the habitat because the wildfires are often uncharacteristically extensive (Lee et al. 1997).

In the basin currently, recreational fishing and to a lesser extent commercial fishing have important economic values. In addition, fish (especially salmon) are extremely valuable to many tribes in the project area for economic, nutritional, and spiritual reasons (McCool et al. 1997). It is estimated that the true effects of wildfire suppression on fisheries resources have appeared only in the past 20 to 30 years, and that they may have only played a small role in the decline of aquatic species. However, it is felt that such effects are becoming more prominent, setting the stage for more severe wildfire effects on aquatic resources in the future (Hann, Jones, Karl, et al. 1997).

Fire suppression has changed the character of vegetation and thereby has contributed to altered timing and volume of stream flow, by changing on-site hydrologic processes (Wright et al. 1990). On rangelands, fire suppression is partly responsible for expansion of western juniper (see earlier discussion), which combined with increasing density can result in decreased understory vegetation (Karl and Leonard 1995); this is believed to contribute to decreased soil infiltration and increased peak discharges during intense rainfall. In forested environments, increased above-ground vegetation due to fire suppression also may have resulted in increased evapotranspiration rates and decreased runoff. Where high intensity fires have increased because of fire suppression, soil porosity has decreased, thus increasing runoff and soil erosion (McNabb and Swanson 1990). Fire can also cause water-repellent layers to form in soils, resulting in temporarily increased runoff (DeBano et al. 1976).

The quality and quantity of water directly influences the lands and resources associated with the rights and interests of tribes, such as instream flows, pools, turbidity. Wildfire suppression has resulted in deteriorating effects on water quality in some parts of the basin especially in the past few years, and the risk is increasing.

Fire Suppression and Air Quality

Clean air and good visibility contribute to the quality of life for people living in the project area and may contribute significantly to the quality of the experience for people who come to the basin to recreate or earn a living (McCool et al. 1997). Wildfires currently have a significant impact on the air resources, degrading ambient air quality and impairing visibility. Because of altered fire regimes due to fire suppression, the area burned by nonlethal understory fires is only one-third of that which burned historically. Stand-replacing fires consume more fuel and produce more smoke than nonlethal fires, which usually burn with fairly low surface fire intensities in the understory. Brown and Bradshaw (1994) found that emissions are greater from current fires, even though they burn fewer total acres than historically, because consumption of fuel per unit area burned has been greater in the current period. Coupled with greater fuel loads in today's forests due to fire suppression, the potential for smoke from wildfires is immense. Inversions during summer are a major cause of the worst ambient air conditions associated with wildfires in the project area.

The effects of poor air quality from wildfire smoke are preventable only through prescribed fire or other fuel reduction activities. Once a wildfire is out of control the smoke is not manageable. Smoke from wildfires can be hazardous to public health, dangerous to travelers, and detrimental to scenic quality.

Fire Suppression and Human Uses

Wildland fire management on Forest Service- and BLM-administered lands will likely become more challenging in the future, because costs of fire prevention and suppression are escalating and effectiveness seems to be declining. Human populations within the urban–rural–wildland interface have substantially increased within the past few decades. These areas of rapidly growing human populations are commonly associated with high fire risk (Hann, Jones, Karl, et al. 1997).

Humans attach emotional, spiritual, and symbolic identification to places—often referred to as a "sense of place," where natural resources are valued not only for functional purposes but also for their value as places to which people, as a community, are attracted and become attached. Altered fire regimes due to fire suppression can have impacts on people by altering the places that have attachment values.

Scenery—the general appearance of a place and the arrangement of its individual features—is another type of amenity provided by federal lands in the project area. Human intervention in natural processes such as fire suppression has been a significant force in shaping visual quality. High scenic integrity is generally considered to occur where native visual qualities are intact and the landscape is unspoiled by human intervention (McCool et al. 1997). As the potential for uncharacteristic wildfire increases in the project area, the potential to reduce visual integrity also increases, since the sight of severe wildfires on the landscape is not considered pleasing to many tourists and residents.

Stand-replacing wildfires that result from past fire suppression often provide an opportunity for salvage harvest in forested environments, which can lead to local economic benefits in the short term. However, such fires decrease harvest opportunities in the long term and reduce the predictability of forest product outputs.

Timber Harvest

Historical to Current Trends

The biophysical environment influences people, their actions, and their systems (social, cultural, economic, political). In turn, people through their actions affect the biophysical environment. Subsequently, human-caused changes in the environment lead to modification of peoples' actions and, potentially, their systems. The continuing cycle represents the working of adaptive management across all scales.

One example of this cycle is the traditional harvest of timber. Euroamerican settlers coming into the basin in the 1800s were initially drawn by the natural environment—an area rich with resources, such as space to live in, raw materials to create shelter and food, and products that could be used to make a living.

Euroamerican settlers brought a cultural system that encouraged domination and use of the natural environment. They brought tools and processes to facilitate that cultural system. They also brought knowledge of and connections to distant markets that had demands for resources from the basin: furs, timber, minerals, meat, and farm products. As a result, within the span of just a few decades, Euroamerican settlers and their descendants had significantly affected the biophysical environment of the basin.

From the late 1800s to the latter 1900s, millions of acres of timber have been harvested. Impacts on the surrounding environment generally were the result of actions to support families and engage in economically rewarding production of goods and services that were desired by the American people. People were directly employed as proprietors or employees in the processes of timber harvesting, milling of lumber, wood products manufacturing, pulp and paper making, use of wood for energy, providing wood to railroads for ties and trestles, providing timbers for mines, and selling other wood products (such as posts, poles, and beams).

As timber harvest businesses proved successful, other people were employed in supporting businesses, trade, and government services (Haynes and Horne 1997). Communities and companies were established and have flourished because of the timber industry. In Idaho, for example, employment of loggers, rafters, and sawmill workers increased from just over 300 in 1880, to more than 8,000 in 1920, to 14,900 in 1995. From 1945 until 1970, timber harvest on federal lands in the project area increased about five percent per year (McCool et al. 1997).

Overview of Timber Harvest Influence

Over the years, timber harvest generated effects in addition to impacts on employment. Timber harvest and forest management practices, along with wildfire suppression, have changed disturbance regimes, natural succession, and vegetation patterns. Roads built to access timber have led to secondary effects, some harmful and some beneficial. For example, roads have caused human disturbance to many terrestrial wildlife species and degraded aquatic species habitat through sedimentation. They have also provided travel routes for other economical and recreational purposes.

The net increase of mid seral forests and the net declines of early seral and late seral forest communities were most likely due to a combination of fire suppression and timber harvest activities throughout the project area. Timber harvest activities reduced the extent of late seral forest communities, while fire suppression activities limited the recruitment of early seral forest communities. Consequently, middle-aged forests now dominate the distribution of terrestrial communities in forested environments more than they did historically.

The rate of change was greater in lower montane than in subalpine forest communities because natural disturbance frequencies within the lower montane were greater than those of the subalpine. Also, successional rates were slower in the subalpine environments. Consequently, the effects of altering disturbance regimes accrued much faster in lower montane environments. In addition, human settlement tended to concentrate in the lower elevation, more hospitable environments. This brought with it the effects of human settlement, development, and uses. Agriculture, urbanization, livestock grazing, fire suppression, and timber harvest in the project area have all had significantly greater impacts over a relatively longer period of time on lower montane forests than on montane and subalpine forests.

Some ecological benefits were derived from timber harvest that otherwise could only have been achieved through a frequent low intensity disturbance. For instance, a harvest thinning can reduce fuel loading and overstocking of trees which, before fire suppression, was accomplished through frequent low intensity fires. These objectives, however, were not often considered important.

From the earliest days of timber harvesting in the project area, the preferred species to harvest were the more valuable shade-intolerant trees such as ponderosa pine, western white pine, and western larch. Of course the largest trees also provided more profit. Of the western white pine cover type that was fairly extensive in the moist forest PVG in the northern parts of the basin at the time of settlement, 95 percent has been eradicated by timber harvest and white pine blister rust.

The result of timber harvest in the low to mid elevation forests is that late seral forests were converted to shade-tolerant mid seral forests through selective harvest of the oldest shade-intolerant trees, or converted to early seral forests through clearcutting or harvest and wildfire. In the case of western white pine, it was often replaced with grand fir, western hemlock, or Douglas-fir. This is an important reason for declines in wildlife species that need late seral forest habitat, such as white headed woodpecker, white breasted nuthatch, or western grey squirrel. It also relates to altered fire regimes, because large shade-intolerant trees are most tolerant of fire and adapted to a regime of frequent, low intensity disturbances.

In the latter part of this century, timber harvest coupled with tree planting greatly speeded up the forest regeneration process, reducing the length of time the forest stayed in the early seral stage. Less early seral forest has meant fewer wildlife species that depend on this habitat, such as Lazuli bunting (Wisdom et al. in press). Reforestation practices have also reduced early seral forest by changing the structure through removal of remaining emergent trees and snags and by shortening the time it takes a stand to move through the stand-initiation stage. Some wildlife species appear to be sensitive to the planting of uncharacteristically high numbers of trees; artificially dense forest stands create unsuitable habitat.

Fire suppression in conjunction with timber harvest allowed mid seral forests to become dense and filled with shade-tolerant species. This combination of management practices also caused the remaining late seral single story forests to develop multiple canopy layers. Early overgrazing by livestock also aided in the process of increased forest stocking in the low to mid elevation forests. One of the effects of increased forest densities has been a reduction in forbs and grasses in the understory used as forage by large game and livestock. On the other hand, large game have found an abundance of hiding cover in these forests. Deer and elk populations have improved with the current mix of habitats in the project area, which has been beneficial for tribes and other people who hunt large animals.

Timber harvest activities have also affected tribes in other ways. American Indian populations in the project area have experienced adverse effects on their way of life and resource uses, some of which was due to timber harvesting. Timber harvest activities and associated roads have produced sediment that has decreased water quality and degraded habitat for salmon and other aquatic species important for economic, nutritional, and spiritual reasons.

American Indians have long used native plants for a wide variety of needs such as food, medicine, incense, lodging materials, and craft materials. Hundreds of native plant and animal species developed cultural importance through subsistence, spiritual, and commercial uses (McCool et al. 1997). These traditions continue today. Increasing forest densities in the low and mid elevations have led to local declines in plant species that are important to tribes such as huckleberry, elderberry, chokecherry, and serviceberry.

Timber harvest can also have direct adverse effects on terrestrial species. The significant declines in old forests throughout the basin and their replacement with mid seral stands, has led to fragmentation of the old forest habitats that remain, which causes a shift in wildlife species composition and vegetation composition. Salvage of dead trees has also reduced the number of snags and amount of downed wood in managed stands.

From settlement time to the present, the amount of soil in the project area disturbed by land management activities has generally continued to increase. Timber removal has at times caused loss of soil organic matter, displacement of topsoil, and compaction of soils, leading to slower infiltration rates, erosion on steep slopes, and disruption of important biological activities. Traditional management activities such as timber harvest, excessive livestock grazing, and roads can directly affect the soil by lowering its long-term productivity, through reducing the soil's capacity to store nutrients and water. This leads to exclusion of fire, invasion of exotics, and development of plant communities on soils that are incapable of supporting increases in biomass. These communities often develop carbon or water stresses and are very vulnerable to wildfire. Nutrients become bound in the woody tissues and are subject to volatilization from severe wildfire. Thus, decline in long-term soil productivity may result.

When uncharacteristic amounts of soil erosion reached streams, it caused degradation of aquatic habitats and resources. Harvest of trees in riparian areas has caused increases in water temperatures. In addition to salmon, many other native fish species have declined since the Euroamerican settlement of the basin, including bull trout, Yellowstone cutthroat trout, westslope cutthroat trout, redband trout, and steelhead. Timber harvest, excessive livestock grazing pressure, uncharacteristic wildfire, water diversions, fishing, introduction of non-native species, urbanization, hydroelectric dams, and agriculture are the causes of these declines.

The increase in insect and pathogen disturbances can be directly correlated with the change in composition, structure, and connectivity of forest host species. Shade-tolerant trees tend to be more susceptible to insects and disease, and the multi-story structures are conducive to spreading infestations. Not surprisingly, causal factors vary geographically by type and intensity of timber harvest, fire exclusion, and subsequently, the resultant stand structures and composition.

Urban–Rural–Wildland Interface Factors

Fire Risk in the Interface

American Indians used fire for their benefit and protection. In low and mid elevation forests, frequent low intensity fires gave fuels little time to build up between fires. Without excessive livestock grazing pressure, rangelands produced enough fuels to support light fires on a frequent basis. Prior to fire suppression activities, fuels were generally maintained at relatively low levels, and areas having high fuel loads were restricted to relatively small isolated patches.

Since settlers imposed fire suppression, live and dead fuels have accumulated throughout much of the forests in the project area. As access to wildlands increased and mechanical equipment and air support became more available, areas burned by wildfire declined through the 1960s. However, by the 1980s, fuel accumulations had generally increased, and areas having moderate to high fuel loadings became larger and more contiguous. In addition, highly flammable noxious weeds were introduced into the rangelands and low elevation forests. Today the occurrence of uncharacteristically large and severe fires has increased substantially.

Human ignitions increased and so did the expansion of urban and rural development into the wildland interfaces. By 1990, the population of the basin had grown to almost three million people, with nearly half the population living in 12 of the 92 counties in the project area. The basin remains far more rural than the U.S. as a whole, but in many areas population growth and development can threaten the qualities that make wildlands attractive for recreation, retirement, and new businesses, particularly because the risk from wildland fire is increasing in urban and rural places alike (McCool et al. 1997). As wildfires become more severe, the associated hazards to life and property will likely increase, as will wildfire suppression costs, making wildfire management on BLM- and Forest Service-administered lands increasingly challenging (Hann, Jones, Karl, et al. 1997).

In low- to mid-elevation forests, urban areas continue to encroach on wildlands even as the fire danger in the forests continue to increase.

In low- to mid-elevation forests, urban areas continue to encroach on wildlands even as the fire danger in the forests continue to increase. Even outside of urban areas, houses, cabins, and small towns are experiencing an increased risk from wildfire. Many forests are becoming more dense, developing multiple crown layers (including smaller trees that create fuel ladders), and converting to shade-tolerant and less fire-adapted species because of past fire suppression, timber harvest, excessive livestock grazing pressure, road building, and other factors (Hann, Jones, Karl, et al. 1997). In rangelands the fire risk stems from increased density of sagebrush and/or juniper in some places, flash fuels created by cheatgrass or other exotic grasses, and other vegetation and fuel accumulation in the absence of grazing. In both forests and rangelands, when fine fuels dry out in the summer, a spark or other ignition source is all that is needed to start a wildfire, and where there is a lot of human activity, there is an abundance of ignition sources.

Smoke is a byproduct of wildfire. When wildfires burn they can create unhealthy levels of smoke in urban areas or other places inhabited by people. Smoke can also put haze in the way of scenery or cloud vistas. When smoke fills the skies, tourists, old people, young people, and those with respiratory problems suffer. Smoke from wildfires is essentially unmanageable. Smoke from prescribed fires, on the other hand, can be created in manageable quantities at times when winds will carry it away from urban area. Smoke from wildfire often comes without warning, while prescribed fires are planned and can be publicized. (See the Fire Suppression and Air Quality discussion, earlier in this section, for additional details.)

Wildlife Conflicts in the Interface

Many species are vulnerable to urbanization because of changes or reductions in available riparian or other habitats, including those brought on by large reservoir construction. Conversion to agriculture and housing development have affected most wildlife species associated with grasslands and shrublands; even though the vast majority of conversions have occurred on private land, their effects are widespread.

Increased conflicts with wildlife have occurred in the urban–rural–wildland interface. Big game species often run into conflict with people and livestock when habitat is reduced or affected by roads. Many urban areas have expanded into the winter ranges of mule deer and elk in recent years, sometimes resulting in animal damage to private property (such as gardens, ornamental plants, or crops) and sometimes causing declines in wildlife populations because of lack of winter forage. Other species such as mountains lions and coyotes are increasing in the urban–rural–wildland interface and causing concern for human safety. These wildlife species are important to tribes, other hunters, and wildlife watchers. Other tribally important wildlife species that have been affected by urban expansion into wildlands include the sage grouse, jackrabbit, and furruginous hawk.

From an aquatic standpoint, urbanization pressures, river channelization, pollution, and other impacts from an increasing human population became evident by the 1960s, as numerous stocks of salmon, steelhead, and sea-run cutthroat trout declined.

Urban expansion into the urban–rural–wildland interface also has affected the availability of traditional plant species to American Indians and their access to those plants.

White Pine Blister Rust

White pine blister rust is a fungal disease that causes branch and stem cankers that often girdle the stem, causing top kill and / or death to the tree. It is the primary introduced disease that has changed successional pathways, cover types, and structures of the cold and moist forest potential vegetation groups.

Large changes from historical moist forest vegetation composition, structure, density, patch and pattern, and disturbance regimes are attributable to the effects of white pine blister rust, harvest activities, fire exclusion, and roads.

Since the settlement of the basin, the western white pine cover type has declined by 95 percent throughout its range in the project area (north Idaho, Washington, and Montana) where there is a combination of climate, abundance of *Ribes* (the alternate host), and susceptible trees (Hann, Jones, Karl, et al. 1997). Another effect of blister rust is poor regeneration of western white pine.

The effects of blister rust go beyond the loss of western white pine, because western white pine fills a unique niche in the ecosystem. Western white pine is a fast-growing, shade-intolerant species that historically depended on fire to remove competing shade-tolerant conifers and to help it regenerate (Graham and Grimm 1990). The loss of western white pine has resulted in a conversion to species such as western hemlock, grand fir, and Douglas-fir. Fire exclusion allowed the forests to become denser, with more canopy layers, smaller trees that create fuel ladders, accumulation of fuels, and trees that retain lower branches. This change has led to less diversity in the understory. It has also changed the disturbance regime to be more severe and reduced the productivity of the ecosystem (Hann, Jones, Karl, et al. 1997). Economically, the residents of the basin have essentially lost a valuable crop to this introduced disease. Wildlife species that need large trees and snags have lost habitat. Examples include the pileated woodpecker, American marten, and northern flying squirrel (Wisdom et al. in press).

White pine blister rust has had a substantial effect in the cold forest PVG as well. Blister rust has reduced the vigor or killed whitebark pine, and the effects are more severe in cold forest than in the moist forest (Hann, Jones, Karl, et al. 1997). The cold climate and short growing season in cold forests slow the natural rate of change in vegetation when compared to dry or moist forests, and the extent of whitebark pine regeneration has declined by 90 percent since the historical period. This leads to concern for high elevation forests of the future.

Hydrologically, loss of scattered whitebark pine trees sometimes causes a disruption of snow pack patterns. Other forest health concerns have been raised because of lower productivity, higher probability of insect and disease infestation, higher probability of high intensity fires, and changes in habitat conditions (Hann, Jones, Karl, et al. 1997).

Some of the wildlife species that need whitebark pine habitat are the Clark's nutcracker and grizzly bear. Grizzly bears in high elevations depend on whitebark pine seeds for a substantial portion of their protein and calories as they build fat to carry them through the winter. Grizzly bears are one of the many species important to tribes.

Roads

Historical to Current Trends

Roads have helped to change the face of the interior Columbia Basin. Some of the earliest roads in the basin were trails traveled by Native Americans. These were expanded by early trappers and prospectors. With the development of the Oregon Trail came a large influx of immigrants to the basin and the beginning of non-native settlement of the project area.

Roads provided the access needed to take advantage of the basin's rich resources: minerals, timber, rangeland, wildlife, fish, recreation, scenery, areas of solitude, and more. Roads have been instrumental not only for the Euroamerican settlement of the basin, but also for the subsequent boom in population, economic expansion, commerce, recreational uses, and the growth and nature of contemporary society. As the number of roads has increased throughout much of the project area, the effects of those roads have also expanded.

Specific Influences of Roads

This section focuses on influences of roads on the ecosystem. Access aspects of roads are described in the Social-Economic-Tribal section , earlier in this chapter.

Streams and Aquatic Species

Roads contribute to the disruption of hydrologic function and increase sediment delivery to streams. Sediment from roads has negatively affected the water and resources dependent on good water quality and quantity, which are important to tribes, communities, recreationists, irrigators, hydropower users, fishermen, and others. The problem is accentuated when roads are old, in sensitive terrain, abandoned, or otherwise not well maintained; roads in conjunction with wildfire add further complications. In general, the closer a road is to a stream, the greater the sedimentation problem.

Roads also provide access for activities such as fishing, recreation, timber harvest, livestock grazing, and agriculture, which have associated effects. In Lee et al. (1997), roads are used as a catch-all indicator of human disturbance on aquatic and riparian systems. Examples of fish species that have declined, in part because of road impacts on aquatic habitat, are Yellowstone cutthroat trout, westslope cutthroat trout, and bull trout (Lee et al. 1997).

Terrestrial Species Habitats

A wide variety of road-associated factors can negatively affect habitats and populations of terrestrial vertebrates as well. Roads are often associated with the effects of human disturbance, which is facilitated by providing access. The density of roads varies greatly across the basin. Existing knowledge about species–road relations was summarized by Wisdom et al. (in press) for the 91 broad-scale species of focus. They identified 13 factors, consistently associated with roads, that can have a negative effect on vertebrates (Table 2-33). At least one of the 13 road-related factors affects over 70 percent of the 91 broad-scale species of focus. In addition, 33 of the 40 groups of species and 11 of the 12 Terrestrial Families have at least one species that is negatively affected by roads.

The negative factors associated with roads are diverse and not always easily recognized. However, several generalizations about effects are possible. Road construction can convert areas of habitat to non-habitat. Roads can provide an avenue for the spread of exotic weeds. Roads create habitat 'edge', which favors species that use edges, often at the expense of species that require more interior habitat. Removal of snags for fuelwood increases along roads. Roads may increase wildlife mortality, through legal and illegal shooting or trapping, vehicle accidents, or poisoning. Roads may increase harassment of species. Roads may restrict movements of small mammals.

Because of these factors, source habitats are probably under-used by many species in areas with moderate or high densities of roads. Furthermore, the negative impacts from roads may exacerbate the negative effects associated with reduction in source habitat. Mitigating the negative effects of roads is challenging, because it requires effective control of human access while balancing societal wants for access and for products (such as recreation, livestock grazing, timber harvest, and minerals) from public lands.

Table 2-33. Road-associated Factors Negatively Affecting Terrestrial Species and Habitats.

Road-associated Factor	Effect of Factor in Relation to Roads
Snag reduction	Reduction in density of snags and/or area where snags are present due to removal near roads, as facilitated by road access.
Downed log reduction	Reduction in density of logs and/or area where logs are present, due to removal near roads, as facilitated by road access.
Habitat loss and fragmentation	Loss and resulting fragmentation of habitat due to establishment and maintenance of roads and road rights-of-way.
Negative edge effects	Specific case of fragmentation for species that respond negatively to openings or linear edges created by roads (such as "habitat-interior" species).
Overhunting	Non-sustainable or non-desired legal harvest by hunting, as facilitated by road access.
Overtrapping	Non-sustainable or non-desired legal harvest by trapping, as facilitated by road access.
Poaching	Increased illegal shooting or trapping of animals, as facilitated by road access.
Collection	Collection of live animals for human uses (for example, amphibians and reptiles collected for use as pets), as facilitated by the physical characteristics of roads or by road access.
Harassment or disturbance at specific use sites	Direct interference of life functions at specific use sites due to human or motorized activities, as facilitated by road access (for example, increased disturbance of nest sites, breeding leks, or communal roost sites).
Collisions	Death or injury resulting from a motorized vehicle running over or hitting an animal on a road.
Movement barrier	Preclusion of dispersal, migration, or other movements as posed by a road itself or by human activities on or near a road or road network.
Displacement or avoidance	Spatial shifts in populations or individual animals away from a road or road network in relation to human activities on or near a road or road network.
Chronic, negative interactions with humans	Increased mortality of animals (such as euthanasia or shooting of gray wolves or grizzly bears) due to increased contact with humans, as facilitated by road access.

Source: Adapted from Wisdom et al. in press, Vol. 1, Table 13.

Exotic Plants

Roads have provided vast avenues of expansion for exotic plants, whose introduction and expansion have drastically affected large areas of habitat. Cheatgrass, for example, cures out in early summer and provides limited forage value compared to native species. Exotic plant expansion has also simplified many plant communities restricting the niches available for many terrestrial animals. The result is loss of productivity, loss of native community structure, loss of native species diversity, loss of habitat, and in extreme cases, changes in the predominant succession/disturbance regimes (Hann, Jones, Karl, et al. 1997).

Vegetation Succession/ Disturbance Regimes

Roads have contributed to the increased departures of vegetation from historical conditions, especially in the lower to mid elevations, where the greatest concentration of roads are located. Through access for timber harvest, livestock grazing, agriculture, and more effective wildfire suppression, road construction has been a partner in the expansion of mid seral shade-tolerant forests at the expense of late seral shade-intolerant forests and to a lesser extent at the expense of early seral forest in the project area. Fire exclusion has also been more effective in roaded areas, consequently changing composition, density, and structure within those areas (Hann, Jones, Karl, et al. 1997). In rangelands, road construction has aided in the expansion of woody species through indirectly assisting fire suppression and livestock grazing. These departures of vegetation have created an imbalance of habitats in the project area: habitat scarcity for those terrestrial species that need late seral forest conditions (such as white headed woodpecker, white breasted nuthatch, and western grey squirrel) and abundance of habitat for those that do well in mid seral forests (such as mule deer and elk; Wisdom et al. in press).

Many forests that have been identified as having forest health problems have been roaded and harvested. In general, wildfires are becoming larger and effects are becoming more uncharacteristically severe because of timber harvest, fire suppression, and roading.

Snags and Downed Wood

There is a high correlation between the high density of roads and the reduction of large snags and coarse woody debris. Snags and downed logs are important components of forest and woodland ecosystems. They provide essential habitat for wildlife, invertebrates, fungi, bryophytes, lichens, and other organisms. They store carbon and nutrients and provide site improvement following extreme disturbance. Snags and coarse woody debris are closely tied, because snags are a future sources of downed logs and coarse woody debris, which recycle nutrients and provide habitat for both plants and animals. Large diameter snags are especially valuable to a wide array of species because they offer greater surface area, more opportunity for cavities, and greater longevity. Hann, Jones, Karl, et al. (1997) found that snag and coarse woody debris levels have declined in roaded and harvested areas.

Tribes

For tribes in the project area, roads have resulted in both positive and negative changes to the current condition of resources important to individual tribes. While roads have increased tribal access to the resources and lands they use, they have also provided greater access to other people. Increased access has led to greater disturbance to cultural and historical resources and increased user conflicts. Recreationists and recreational uses of public lands have increased, resulting in growing conflicts with tribal uses, some of which are referenced and guaranteed under treaties (such as fishing, hunting, trapping, and gathering). Examples of recreational or commercial-type uses which can conflict with tribal uses or which can negatively affect resources important to tribes are: commercial raft and float trips, 'new-age' activities, rock climbing, berry picking, and gathering of basketry or ornamental plant materials. In some cases, the recreational activity has changed the condition of the resource/site important to tribes—for example, rock climbers putting pitons in a mountain or rock traditionally used by a tribe, or more visitors affecting dispersed camping sites or back country trails commonly used by a tribe.

Tribes are sometimes being out-competed by commercial operations for resources important to their rights and interests. There has been an increase in the commercialization of many plant species (for example, beargrass, mushrooms, huckleberries). Commercial pickers are increasingly competing with tribes for these resources, many which are associated with the exercise of a treaty reserved right.

Biophysical changes brought on by roads are also of concern to the tribes because changes to project area forests, rangelands, riparian areas, or streams affect American Indians' ability to obtain traditional plants, wildlife, fish, and cultural objects. For instance, as the Snake River sockeye and spring and fall chinook salmon runs have declined, so have the diets, economies, and culture of basin tribes. Wildlife species that have been negatively affected by roading and increasing human disturbance include grizzly bears, grey wolves, and other large carnivores.

Livestock Grazing

Grazing Before Euroamerican Settlement

Vegetation throughout the project area was in a continued state of change during the late Pleistocene (roughly 132,000 to 10,000 years ago; Grayson 1993) and the Holocene (the last 10,000 years; Grayson 1993, Miller et al. 1994). Changes in climate and frequency of fire were probably the primary disturbances influencing vegetation change before Euroamerican settlement. Grazing played a less dominant role (Aikens 1986, in Miller et al. 1994; Grayson 1993; Martin 1967, in Miller et al. 1994; Miller et al. 1994).

Large grazers were more abundant and diverse during the Pleistocene compared with the Holocene (Allison 1996, in Grayson 1993; Howe and Martin 1967, in Grayson 1993; Grayson 1993). Many of the large grazers, and various other mammals, became extinct in the late Pleistocene (Grayson 1993) but undoubtedly were encountered by the earliest human occupants of the Great Basin (Grayson 1993).

Large grazer abundance and diversity, at least in the Great Basin portion of the project area, were apparently much reduced during the Holocene compared with the late Pleistocene (Grayson 1993). Elk were present in the Great Basin portion of the project area at the end of the late Pleistocene and during the early Holocene (roughly 11,000 to 7,200 years ago; Grayson 1993). However, elk abundance declined substantially after this period. Grayson (1993) maintained that the Great Basin supported a far richer assemblage of large mammals toward the end of the Pleistocene than it does at current.

Grazing pressure thus probably declined in the Holocene. In the late Holocene, before Euroamerican settlement, environmental conditions along with hunting pressure by American Indians appeared to keep large grazer numbers low. Grazing impacts by large grazers were probably light, except in localized areas (Miller et al. 1994). Grazing was probably seasonal, with animals migrating between upper-elevation summer range and lower-elevation winter range (Burkhardt 1996, Miller et al. 1994). Consequently, the effects of grazing by large grazers was probably minimal and did not cause vegetation changes (Miller et al. 1994).

Livestock Grazing Since Euroamerican Settlement

Since the historical period, grazing pressure has increased substantially as Euroamericans introduced horses, sheep, and cattle. In the *alpine PVG,* excessive grazing pressure, primarily by sheep, has caused excessive erosion and removal of alpine vegetation (Hann, Jones, Karl, et al. 1997). In the early 1900s, Griffiths (1902, 1903) reported declines in native vegetation abundance and condition in the Great Basin portion of the project area and in eastern Oregon and Washington. He attributed this decline to the introduction of domestic livestock (particularly cattle and sheep) and wild horses to the region, and excessive livestock grazing pressure. The excessive livestock grazing pressure and its adverse effects on native vegetation were particularly apparent on Steens Mountain in southeast Oregon. Grassy areas, riparian areas (characterized by willow), and areas with aspen, were particularly overused by livestock. Griffiths estimated that in 1901, 182,500 sheep, or 175 sheep per square kilometer, were present on Steens Mountain.

Historical evidence indicates that most *riparian areas* in the project area have changed dramatically in the past 100 years, attributable greatly to excessive livestock grazing pressure (“improper livestock grazing” in Chaney et al. 1990). Improper livestock grazing was defined by Chaney et al. (1990) as concentrations of livestock at the wrong time (that is, season), in too great a number (that is, intensity), for too long (that is, duration), or any combination of these factors that results in riparian area degradation.

The change in riparian areas due to livestock grazing is related to livestock distribution and behavior. While riparian areas constitute only a small percentage (two to three percent) of the project area (Hann, Jones, Karl, et al. 1997, table 3.18), livestock (particularly cattle) activity has been disproportionately concentrated within riparian areas (Kovalchik and Elmore 1992, Marlow and Pogacnik 1986) compared with upland areas. Concentrated livestock activity in riparian areas has resulted in excessive grazing and physical damage by trampling.

Some ramifications of this include: (1) an increase in stream energy, (2) more bare soil and accelerated erosion, and (3) stream channel degradation, which has resulted in less water recharge of floodplains, lowered water tables, and reduced geographic extent of riparian plant communities. Erosion and stream channel degradation have caused an increase in suspended sediments and declines in water quality. Water temperatures have increased because of a decline in shade provided by vegetation. The structural diversity of vegetation has been simplified, early successional species have increased in abundance, and the result has been less productive plant and animal assemblages. The decline in abundance of some riparian species such as willow and cattails, which are associated with the rights and interests of tribes, has had a negative impact on tribes. Direct influences of livestock concentrations in riparian areas on water quality include bacterial and protozoal parasite contamination, and nutrient enrichment from fecal material in and near surface waters (Larsen 1996).

Livestock grazing has played a role in the *dry forest PVG* as well. Before Euroamerican settlement of the project area, and before extensive livestock grazing was introduced to the region, ponderosa pine forests were typically savannah-like, appearing open and parklike, with stands of grass of varying densities and sparse tree recruitment (Coville 1898, Leiberg 1899, Leiberg et al. 1904, Pearson 1923, Rummell 1951, several historical accounts cited in Cooper 1960, Gruell et al. 1982). Livestock grazing, particularly excessive livestock grazing pressure (Weaver 1947b, Arnold 1950, Rummell 1951), has been implicated as one factor of many that have stimulated a shift from open parklike stands to stands with greater density of trees. On some areas within these forests, excessive livestock grazing pressure resulted in decreased herb abundance in the understory, which caused a decrease in fine fuel loads and a reduction in fire severity, which promoted tree seedling establishment.

In the late 1800s, livestock grazing played a major role, along with agriculture and the development of the railroad system, in the establishment and spread of *exotic undesirable plants*, many of which are now legally declared noxious weeds. Livestock grazing then and now acts to spread exotic undesirable plants, such as cheatgrass, medusahead, halogeton, and many others. Seeds of these species can be spread through livestock feces, fleeces, and hooves, and many pass through the digestive system and still retain their germination ability (Stoddart et al. 1953, in Mack 1986; Mack 1986). In addition to livestock, native grazers such as mule deer and elk, and birds such as mourning doves, perform this same role of seed spread of exotic undesirable plants.

Grazing pressure by large grazers on native vegetation in the project area has thus fluctuated over time. Large grazers were more abundant and diverse during the Pleistocene epoch compared with the Holocene epoch, and grazing pressure probably declined in the Holocene. Grazing pressure increased substantially as EuroAmericans settled the area and introduced sheep, cattle, and horses, perhaps even greater than during the Pleistocene. Adverse effects on native vegetation, particularly during the late 1800s and early 1900s, were evident in riparian areas, upland rangeland vegetative types, and the ponderosa pine and mixed-conifer forests of the dry forest PVG. These effects have resulted in many subsequent effects, such as increasing noxious weed invasion and spread, and negative effects on the plant species associated with the rights and interests of tribes. Grayson (1993) evaluated these changes from historical to current attributable to livestock grazing and concluded that, for the grasses, "The native grasses of the floristic Great Basin are not adapted to *heavy* [italics added] grazing by large mammals."

Interrelationships of Livestock Grazing and Vegetation Change (Succession) from Historical to Current

Platou and Tueller (1985) proposed that livestock grazing, and livestock grazing systems, that were patterned similarly to grazing as it happened pre-Euroamerican settlement, would be more compatible with vegetation in the Great Basin. However, pre-Euroamerican settlement conditions no longer exist in the project area. Agricultural and urban development, livestock grazing, the introduction of exotic plants, changes in climate (Tausch 1998), and changes in disturbance frequencies and severities, have resulted in unprecedented changes. Given these changes, two pertinent questions can be asked regarding livestock grazing.

Models of Vegetative Succession

Question 1: Can ecosystem functions and processes (such as vegetative succession) that have been altered by excessive livestock grazing pressure since the historical period be restored by removing livestock?

Answer: 1. Yes, in wetter systems such as riparian vegetative types. 2. Yes, in drier systems such as upland rangeland vegetative types, within areas that have not crossed a threshold to a stable, lower successional state. 3. No, in drier systems such as upland rangeland vegetative types, within areas that have crossed a threshold to a stable, lower successional state. See discussion below for more detail.

Current scientific thinking regarding livestock grazing pressure and its relation to vegetative succession typically falls into two general categories of models (Laycock 1994). The first and older model of vegetative succession is the traditional ***"climax" model*** (Figure 2-22), based on the work of Clements (1916) as modified for rangelands by Sampson (1919). The climax model is essentially a model upon which range condition is assessed, labeled typically as excellent, good, fair, or poor.

As used, the climax model has three assumptions:

1. A vegetative type has only one stable state, the climax, which is a stable plant community determined by climate.
2. Any change in the plant community away from climax (which is referred to as retrogression) that is caused by excessive livestock grazing pressure, results in an unstable state that can be reversed by reduction, manipulation, or elimination of livestock grazing. This reversal represents a movement of the plant community back towards the climax community, which is secondary succession. Thus, retrogression and secondary succession are opposite pathways of vegetation change; retrogression leads vegetation away from climax and thus into poorer condition, and secondary succession leads vegetation toward climax or excellent condition.
3. For a given plant community, its condition can change from poor to excellent or from excellent to poor. The change is continuous along a continuum (Vavra et al. 1994).

While livestock grazing management in the project area has been guided by principles of the climax model of vegetative succession during the 20th century, rangeland scientists have accumulated convincing evidence that not all rangeland vegetative types respond according to the climax model. Those that fit the climax model the best are the riparian PVGs, the cool shrub PVG, and the dry forest PVG. There are exceptions even within these PVGs where improvement might not be detected, particularly in cases of extreme past grazing abuse and / or noxious weed invasion.

The second, and more recent model is the ***"state and transition" model*** (Figure 2-23). This model is being proposed as more operative for most arid and semi-arid vegetative types in the interior West (Tausch et al. 1993, Laycock 1994, Tausch 1998). The reason is because many rangeland vegetative types, if they have retrogressed to lower successional states, can remain stable at these lower (more degraded) successional states for long periods of time, even if livestock grazing pressure has been reduced or eliminated. Under these conditions, active restoration in the form of rangeland modifications such as seedings and weed control are needed in addition to reduction or elimination of livestock grazing pressure to achieve secondary succession. Some examples of these vegetative types include ones in the dry shrub PVG, such as the big sagebrush, low sage, and salt desert shrub cover types.

The state and transition model defines vegetative states as recognizable, relatively stable assemblages of species occupying a site. Disturbances such as

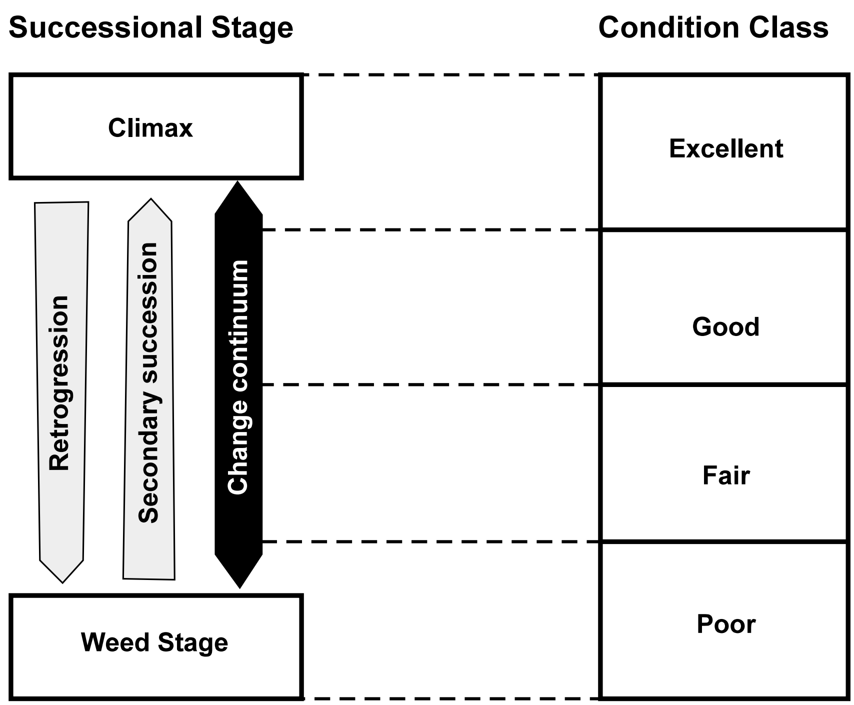

Figure 2-22. Climax Model For Vegetation Succession.
The climax model of vegetation succession and the approximate relationships between range condition and degree of retrogression from climax conditions. (Adapted from Ecological Implications of Livestock Herbivory in the West.

excessive livestock grazing pressure or altered fire frequency and severity, can cause vegetation to cross a threshold, or transition, to a stable state. These stable, lower successional states are typically not desired. Examples of apparently stable vegetative states in the project area are the cheatgrass–mustard and / or medusahead dominated areas within the dry shrub PVG, prevalent in the Lower Snake RAC, and western juniper dominated areas within the cool shrub PVG, observed in the Deschutes PAC, John Day RAC, and Lower Snake RAC.

In the cheatgrass–mustard and / or medusahead example, frequent fires result from the establishment of these highly flammable exotic annual grasses. Current fire-return frequencies as low as five years largely prevent establishment of perennial grasses and shrubs. Removal of livestock will not reduce fire frequencies and can exacerbate fire susceptibility through the subsequent accumulation of flammable litter.

The western juniper stable state is an example of the encroachment of woodland cover types and structural stages into the cool shrub and dry grass PVGs. Between historical and current periods, excessive livestock grazing pressure and fire suppression were two main factors that caused this encroachment. Excessive livestock grazing pressure, particularly in the late 1800s and early 1900s, contributed to a reduction in fuels that could carry fire, thereby decreasing fire frequency. Because woodland species, such as western juniper, can be killed from fire, a decrease in fire frequency favored their persistence and spread. In addition, excessive livestock grazing pressure, through consumption of herbaceous species, contributed to an increase in density and canopy cover of shrub species, primarily sagebrush. The establishment of shrubs provides conditions favorable to establishment of such woodland species as western juniper. Reduction or elimination of livestock grazing pressure will not necessarily convert dominance by woody plants to dominance by grasses and forbs, particularly on sites with these stable states where woody plant cover is dense and there is a sparse grass and forb understory. However, adjustments in livestock grazing pressure or rest from livestock grazing can result in improved soil stability (perhaps through biological crust development, for example), soil water levels, and nutrient levels, particularly on sites that have yet to cross the threshold to the stable vegetative state (Archer 1994; Hann, Jones, Karl, et al. 1997).

Estimates of the extent of rangeland vegetative types on BLM and Forest Service administered lands that have (1) crossed a threshold to a lower and more degraded stable successional state, (2) have not crossed a threshold but are at imminent risk of doing so, and (3) have not crossed a threshold and are not at imminent risk of doing so, are unknown. Knowing this information would help land managers determine the extent of rangelands that are in need of restoration, the intensity of restoration activities that would most likely achieve restoration, and the level of risk associated with achieving the restoration. With this infromation land managers could identify those rangelands that are in need of restoration that would most likely respond positively to changes in livestock grazing management alone, and identify those rangelands that are in need of restoration that would likely require costly, active resoration in addition to changes in livestock grazing management.

Livestock Management and Native Vegetation

Question 2: Can livestock be managed in a manner that would be compatible with native vegetation in the project area?

Answer: Yes, given certain conditions. See discussion below for more detail.

Archer and Smeins (1991) identified several examples of poor compatibility of livestock grazing management practices with native vegetation:

1. Traditionally, livestock are concentrated at artificially high levels. In contrast, densities of native grazers varied by season and by year.
2. Fences prevent livestock from moving to new areas when the abundance of desired forage declines. Consequently, traditional grazing practices result in higher frequencies and intensities of grazing than would have occurred with pre-Euroamerican settlement grazing.
3. Mortality of native grazers was a feedback loop that reduced grazing pressure, permitting recovery of native vegetation after periods of forage overuse. Supplemental feeding precludes mortality of livestock and maintains grazing pressure over a greater portion of the year and over a higher frequency of years, compared with grazing pressure exerted by native grazers.
4. As noted previously, prolonged grazing in grasslands or woodlands that are capable of supporting trees and shrubs has decreased the capacity of grasses to competitively exclude woody plants. It also concurrently reduced fire frequency, and usually, fire intensity, by preventing the accumulation of fine fuels. This has led to the western juniper stable state discussed previously.

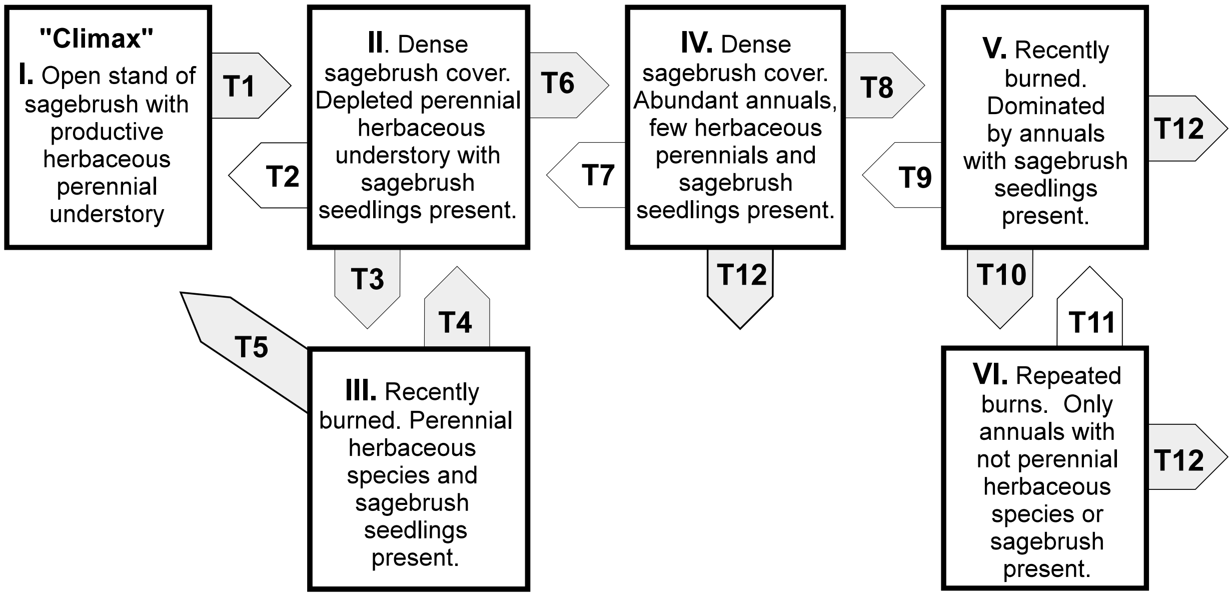

Figure 2-23. State and Transition Model for Sagebrush Grass Ecosystem.

States I, II, and III exist in areas without annual species (for example, cheatgrass or medusahead).

State I is the "climax" or condition undisturbed by livestock grazing. Transition arrow T1 represents heavy grazing which causes deterioration of the understory and increased density and vigor of sagebrush.

State II is dominated by sagebrush and will remain stable for long periods of time. Transition T3 is fire or some other force (for example, insects, disease, or an herbivore that eats sagebrush) that reduces the sagebrush, which permits the understory to improve (State III).

With proper livestock grazing management (Transition T5), State III can move back to a state resembling State I. With heavy grazing (Transition T4), State III will move to State II, and sagebrush will again dominate the stand. State IV represents the situation in a heavily grazed area where a well-adapted annual-like cheatgrass exists. Continuous heavily grazing (Transition T6) of State II results in State IV, and perennials in the understory have been replaced by annuals.

The transitions of State IV to State V (Transition T8), and State V to State VI (Transition T10) represents the role of fires in the conversion to a stable cheatgrass-dominated plant community. Transition T12 represents intervention by humans, such as seeding of exotic perennial grasses (for example, crested wheatgrass). The Bureau of Land Management, for example, plants strips of vegetative fuel breaks consisting of crested wheatgrass, other grasses, forbs, and shrubs to slow the spread of fires. (Adapted from Ecological Implications of Livestock Herbivory in the West).

Grazing systems have been promoted to mitigate or prevent the detrimental effects on native plant communities. Under specific circumstances, rest-rotation, deferred, deferred rotational, and seasonal grazing systems have all been demonstrated to sustain upland rangeland plant communities within the sagebrush grassland and pine–bunchgrass zones in the project area (Vallentine 1990). However, none of these grazing systems have been conclusively more effective than light-to-moderate stocking levels under continuous seasonal use (Hart and Norton 1988, Heady 1975, Stoddart et al. 1975, Vallentine 1990). Thus, despite the array of grazing systems conceived and promoted during the past 40 years, there has been considerable debate over their compatibility with upland native plant communities. The debate is focused on which grazing system(s) are best prescribed to achieve compatibility with specific native plant community(ies), rather than on whether or not grazing systems as a whole are compatible.

In riparian areas, while total exclusion of livestock will improve riparian area conditions (Claire and Storch 1977, Duff 1977, Gunderson 1968, Winegar 1977), total exclusion is not always necessary to reduce negative impacts (Krueger and Anderson 1985). Land managers have been able to accomplish riparian area improvement concomitantly with livestock grazing (Chaney et al. 1990, Elmore 1992, Elmore and Kauffman 1994) through an increased emphasis on compliance with suitable grazing strategies and practices. There are limitations, however, associated with livestock grazing because, “In essence, livestock are NOT a ‘tool’ to improve riparian ecosystems. Rather, they are a cost that may often be accommodated and still enable successional advancement of riparian vegetation and attendant functional values [Krueger and Anderson 1985, Kindschy 1987]” (Kindschy 1994).

Grazing strategies, grazing practices, and grazing systems that are beneficial to achieving riparian area improvement can be found in more detail in Hann, Jones, Karl, et al. (1997). Numerous case study examples of riparian area improvement in the project area stem from incorporation of these grazing strategies, practices, and systems (Chaney et al. 1990, 1993; Kinch 1989). For many of the successful case studies, exclusion of livestock (two years or more) jump-started the recovery, thereby enhancing the effects of improved management implemented thereafter.

In summary, livestock grazing can be managed to sustain and even improve riparian vegetative types. Livestock grazing also can be managed to sustain upland rangeland vegetative types. However, particularly for the drier rangeland plant communities, when they have crossed a threshold and transitioned to a lower successional stable state, grazing systems, and no grazing, are unlikely to achieve a transition to a higher successional stable state (Archer and Smeins 1991). Sustainable grazing management, then, relies on knowledge of critical thresholds and manipulation of livestock (use of appropriate grazing systems, strategies, and practices) so these critical thresholds are not exceeded. Continued stocking at near-normal levels during periods of moderate to severe drought is probably the greatest cause of range deterioration (Vallentine 1990) and crossing of critical thresholds. Vallentine (1990) proposes that reduced livestock grazing intensities during moderate to severe drought, and for some time after drought, are necessary to minimize damage and hasten recovery of perennial vegetation.

Noxious Weeds and Other Exotic Undesirable Plants

The project area has experienced numerous exotic plant invasions in the past 100 years ([Franklin and Dyrness 1973, Yensen 1981, and Young and others 1972] in Mack 1986). As of the mid 1990s, approximately 862 species of exotic plants existed within the Pacific Northwest (Washington, Oregon, Idaho, Montana, and Wyoming; Rice 1994), nearly all of which inhabit the project area. These 862 exotic plant species represent 43 percent of the estimated 2,000 exotic plant species present in the entire United States (U.S. Congress, Office of Technology Assessment 1993, in Vitousek et al. 1996).

Many of the exotic plants existing within the project area originated in the Mediterranean region. The climate of the Mediterranean region (wet, cool autumns and winters; and dry, hot summers) is similar to the climate of the project area. Thus, many exotic plants are adapted to the project area climate (Trewartha 1981, in Mack 1986; Young et al. 1972, in Mack 1986).

Euroamerican settlement of the project area in the late 1800s facilitated the invasion and spread of exotic plants. Agriculture was the major avenue by which exotic plants initially entered the project area. The seed of many exotic plants was a contaminant of crop seed. The land-use change from wildlands to agriculture—a transition that was the most prevalent change between the historical and current periods in the project area—has promoted invasions of numerous exotic plants.

Of the 862 exotic plant species existing within the five-state region, 115 have been legally declared as "noxious weeds" by at least one of the five states.

Of the 862 exotic plant species existing within the five-state region, 115 have been legally declared as "noxious weeds" by at least one of the five states. "Noxious" is a legal classification and not an ecological term. Plants that can exert substantial negative environmental or economic impact can be designated as "noxious" by various governmental agencies. Noxious weeds are therefore a subset of the exotic plant species.

Success of Exotic Plants–Noxious Weeds

Present distributions of many exotic plants within the project area, including the noxious weeds, are increasing rapidly and in some cases exponentially (Asher 1994, Rice 1994, Rice and Rider 1995). This rapid rate of expansion has overwhelmed the ability to curtail the expansion. Uncoordinated weed control efforts throughout the project area have been ineffective against noxious weeds and other exotic plants.

This rapid rate of expansion is partly due to the life history of exotic plants. They are frequently among the first species to arrive and colonize areas where the soil surface has been disturbed or where plant cover is lacking. Their establishment and spread is aided by disturbance to the soil surface (Baker 1986, Bazzaz 1986). Exotic plants that have an opportunistic, colonizing life history—referred to as "colonizers" (Bazzaz 1986)—are typically prolific producers of seeds (or other reproductive parts such as rhizomes) and often are adapted to long-distance dispersal by means of vehicles, wind, wildlife, livestock, water, or machinery. They usually germinate under a wide variety of conditions, establish quickly, grow fast, and out-compete native species for water and nutrients. Some of the densest infestations of exotic plants are near roads, which provide a route for spread.

Other exotic plants, such as the noxious weeds spotted knapweed, yellow starthistle, and leafy spurge, can be labeled "invaders" (Bazzaz 1986). Invaders can establish within relatively intact vegetative cover, and displace native species without the aid of soil-surface disturbance. While noxious weeds can be colonizers or invaders or both, depending on the vegetative cover type, it is noteworthy and perhaps indicative of their noxious weed status that many of them act as invaders. For example, spotted knapweed, yellow starthistle, and leafy spurge have the ability to invade relatively undisturbed sites, including wilderness areas and national parks (Asher 1994, Tyser and Key 1988).

Why Exotic Plants–Noxious Weeds are a Problem

The rapid expansion of exotic plants–noxious weeds in the project area is one of the greatest threats to healthy native plant and animal communities. Noxious weeds are reducing the value of these native plant and animal communities in several ways, including: (1) decline in quality of aquatic-riparian and terrestrial habitats for wildlife; (2) reduction of forage for grazing animals; (3) potential increase in water runoff, sediment delivery, and soil erosion; (4) potential decline in water quality; (5) reduction in biological diversity; (6) negative impacts on or declines in native plant resources associated with the interests or reserved rights of American Indian tribes (see Appendix 8 for a partial list of these plants); and (7) increase in the economic burden of maintaining the quality of recreation and wilderness areas.

The invasion and spread of exotic plants can change the structure and composition of vegetative cover types and can change succession, preventing succession from leading to the vegetation that is the potential for a site. Indeed, the invasion and spread of exotic plants such as cheatgrass, medusahead, and many noxious weeds is apparent within many lower (less advanced) and relatively stable successional states in many rangeland vegetative cover types (see state and transition model of succession discussion in Livestock Grazing section previously). Native plant cover types can be changed to an exotic cover type. The reduction in biodiversity from the site scale to the watershed scale is becoming reality now. Billings (1994) warns that in the cheatgrass-dominated areas of the Intermountain region, including the Snake River Plains of the Lower Snake RAC, some native species are in danger of extirpation at the local or regional scale (see cheatgrass discussion below for more information on cheatgrass).

Accumulating evidence is revealing that invasions of exotic plants into native plant cover types can increase surface runoff and sediment yield (Lacey et al. 1989). This suggests that exotic plant species are not "holding the soil" as well as native species.

The susceptibility of vegetative cover types to invasion by noxious weeds and other exotic plants (see discussion on Susceptibility below for more detail) has led to declines in geographic extent of several vegetative cover types in the project area between the historical and current periods. Table 2-34 shows some selected cover types that have declined between the historical and current periods, partly because of invasion by the noxious weeds listed (see Appendix 5 for more detail). These cover types are important source habitats for the terrestrial vertebrates in the 12 Terrestrial Families.

Susceptibility of Broad-scale Vegetative Cover Types to Invasion

Dewey and et al. (1991) propose that "The precision and usefulness of federal weed control Environmental Assessment (EA) and Environmental Impact Statement (EIS) documents would be significantly improved by knowing the exact location and extent of lands vulnerable to specific noxious weeds." To this end, a measure of the susceptibility of the broad-scale vegetative cover types in the project area, to invasion by 25 weed species (24 noxious weeds, plus cheatgrass) is presented in Tables 2-35 and 2-36. For detailed discussion of each of the 25 weed species, including their county distribution within the project area and which cover types are susceptible to them, refer to Hann, Jones, Karl, et al. (1997).

Some major findings from Table 2-35 include the following. The first five findings are in agreement with Baker (1986) and Forcella and Harvey (1983).

- Grasslands, riparian areas, and some relatively open forests are more susceptible to invasion by exotic plants than are dense forests, high montane areas, and deserts. The former have frequent gaps in the plant cover, which favor exotic plant establishment, whereas the latter have relatively closed plant cover or have extreme climate, which is tolerated by only a few exotic plant species.

Table 2-34. Vegetative Cover Types in Decline Because of Noxious Weeds and Exotic Plants.

Cover Type[1]	Associated Potential Vegetation Group[2]	Noxious Weeds–Exotic Plants
Wheatgrass Bunchgrass	Dry Grass	Diffuse knapweed, spotted knapweed, yellow starthistle, rush skeletonweed, sulfur cinquefoil, medusahead, Dyers woad, dalmatian toadflax, yellow toadflax, common crupina
Fescue-Bunchgrass	Dry Grass	Spotted knapweed, leafy spurge, sulfur cinquefoil, oxeye daisy
Antelope Bitterbrush-Bluebunch Wheatgrass	Dry Shrub	Diffuse knapweed, cheatgrass[3], dalmatian toadflax, rush skeletonweed, sulfur cinquefoil
Big Sagebrush	Dry Shrub	Cheatgrass, medusahead, diffuse knapweed, rush skeletonweed, dalmatian toadflax, Dyers woad, Mediterranean sage, yellow starthistle
Herbaceous Wetlands	Riparian Herb	Kentucky bluegrass[2], Canada thistle, purple loosestrife, leafy spurge, saltcedar, musk thistle, Russian knapweed, spotted knapweed, Scotch thistle, yellow starthistle, hoary cress (whitetop), Mediterranean sage
Shrub Wetlands	Riparian Shrub	Canada thistle, leafy spurge, musk thistle, purple loosestrife, saltcedar, Russian knapweed, Mediterranean sage

[1] Selected vegetative cover types in the project area that have declined in area from historical to current periods, in part because of the noxious weeds listed for each type.
[2] The associated potential vegetation group in which the cover type resides.
[3] Not legally declared noxious in project area.

Source: Karl et al. (1995)

- The exotic forbs / annual grass cover type is the most susceptible to invasion by exotic plants. All 25 exotic plant species show some affinity for this cover type.
- Except for the exotic forbs / annual grass cover type, the grassland cover types (particularly fescue-bunchgrass, herbaceous wetlands, and wheatgrass bunchgrass) are the most susceptible to invasion by exotic plants. This finding is based on the large number of exotic plants labeled "invaders" in these grassland cover types.
- High-elevation cover types, particularly alpine tundra, whitebark pine / alpine larch, and whitebark pine, are the least susceptible to invasion by exotic plants. This finding is based on the small number of exotic plants labeled "colonizers" or "invaders" in these high-elevation cover types.
- Moist and shady forested cover types with little light in the understory (such as grand fir / white fir, mountain hemlock, Pacific silver fir / mountain hemlock) appear to be less susceptible to invasion by exotic plants than are drier, open-canopied forested cover types with greater light in the understory (such as interior ponderosa pine).
- Extremely arid cover types are less susceptible to invasion by exotic plants. For example, of all the rangeland cover types, salt desert shrub is the most arid and is also one of the least susceptible to exotic plant invasion.
- Some exotic plants show no affinity to many cover types in the project area. For example, some species such as purple loosestrife are found only in riparian areas.

Table 2-35 is a risk index that permits land managers and the concerned public to assess which cover types are most at risk of invasion by exotic plant species. Table 2-35 will require further revision as more information becomes available. For example, locality records for exotic plants including information on plant species that were found in the vicinity, would provide a link between the exotic species and a cover type and improve the ratings given in the risk index. Improvement in this risk index will enhance the ability of all to predict risk of invasion, assess where loss of biodiversity is at greatest risk, and assess where risk to changes to succession are greatest. Predicting noxious weed distributions in the future requires that we know which vegetative cover types are susceptible to invasion by the weed species, and where these cover types exist in relation to where the noxious weeds are distributed currently.

Integrated Weed Management

The least expensive, most effective, and highest priority weed management technique is prevention, especially prevention of new infestations of existing noxious weed species, and prevention of establishment of new exotic plants not currently residing in the project area. The magnitude and complexity of noxious weeds in the project area, combined with their cost of control, necessitates using Integrated Weed Management (IWM). Integrated Weed Management highlights the importance of prevention and involves the use of several control techniques in a well-planned, coordinated, and organized program to reduce the impact of weeds. The IWM strategy is discussed in more detail in Appendix 11.

Cheatgrass

Cheatgrass is an annual grass that was introduced to the project area from Europe in the late 1880s, probably via contaminated grain (Mack 1981, Mack and Pyke 1983). By 1930, cheatgrass had already attained its current distributional range in the western United States (Mack 1981) and has since been increasing in density. In 1995, cheatgrass existed in every county in the project area (Karl et al. 1995). A strong case could be made that cheatgrass is the most abundant exotic plant in the project area.

Cheatgrass has adapted to many cover types, from low-elevation salt desert shrub (Sparks et al. 1990, Young and Tipton 1990) to higher elevation ponderosa pine cover types (Daubenmire 1952). These cover types exist at elevations ranging from about 1,477 to 9,000 feet (450 to 2,745 meters), where the annual average precipitation ranges between 6 and 22 inches (15 and 56 centimeters; Bradley 1986).

Cheatgrass has several characteristics that aid its establishment in native plant cover types, particularly cover types that are under stress or have been disturbed. These characteristics include high seed production (Hulbert 1955), ability to germinate in the autumn or spring, greater ability to germinate than native grasses (Mack and Pyke 1983, Martens et al. 1994), tolerance to grazing, and population increase attributable to frequent fire (Klemmedson and Smith 1964).

Standing dead cheatgrass and litter produced by cheatgrass is extremely flammable and causes more frequent fire compared with fire frequency of the pre-

Euroamerican settlement period (Billings 1948). Native sagebrush cover types had fire-return intervals of 32 to 70 years (Wright et al. 1979), whereas cheatgrass-dominated areas that used to be sagebrush now burn as frequently as every 5 years or less (Pellant 1990). This situation is referred to by Pellant (1996) as the “cheatgrass–wildfire cycle”. As a result of cheatgrass invasion and dominance and the more frequent fires, the extent of big game winter range in the Great Basin has declined (Pellant 1990, Updike et al. 1990), habitat supporting the densest concentration of nesting raptors in North America has declined (Kochert and Pellant 1986), the persistence of some native plant species is threatened (Rosentreter 1994), native plant species diversity has declined, non-game bird abundance has declined (Dobler 1994), and succession to the potential vegetation has been slowed or stopped (Billings 1994, Whisenant 1990). The cheatgrass–wildfire cycle presents the greatest risk to the Wyoming big sagebrush areas of the big sagebrush cover type, and to the wetter portions within the salt desert shrub cover type (Pellant 1990, Peters and Bunting 1994).

Cheatgrass typically provides adequate soil surface cover for watershed protection. However, in drought years and after wildfires, cheatgrass production can be inadequate to provide soil surface cover suitable for watershed protection. This is especially evident on sites with soils susceptible to water and wind erosion, and on sites with moderate to steep slopes. Under these circumstances, the potential for erosion is greater.

Ecological relationships between cheatgrass and biological crusts are not understood completely. Where intact, biological crusts apparently can restrict cheatgrass establishment (Kaltenecker and Wicklow-Howard 1994), but biological crust development appears to be restricted within cheatgrass-dominated plant communities, in comparison with native plant communities (Pellant and Kaltenecker 1996). After burning, cheatgrass can rapidly dominate sites and hinder the recovery of biological crust species. The lack of biological crust development and species richness might have negative implications in nutrient cycling, native plant succession, site stability, and exotic species invasion. Biological crusts are discussed in more detail in the Terrestrial Species section of this chapter.

Cheatgrass may be controlled by mechanical (disking or plowing), burning, grazing, herbicides, and biological control methods. These techniques vary in their effectiveness depending on factors including the growth stage of cheatgrass at the time of application, pre-and post-application climatic conditions, and soil water. Following control, revegetation with perennial plants is normally necessary.

Research conducted in southern Idaho (Hull 1974, Hull and Holmgren 1964, Hull and Stewart 1948) provides strong evidence that introduced wheatgrasses, especially crested wheatgrass, are superior to native grasses in establishing and persisting in communities previously infested with cheatgrass. However, controversy surrounds the use of exotic plants to revegetate rangeland communities infested with cheatgrass because of the resulting reduction in native plant species richness. Post-wildfire seeding with seed mixtures composed primarily of crested wheatgrass was a common practice to prevent cheatgrass dominance after wildfires and to provide livestock forage from the 1950s to the 1970s. This practice has continued to a certain extent into the current period (Pellant and Monsen 1993), and such seedings have reduced the extent of cheatgrass monocultures on the landscape. Although seeding of perennial grasses tends to perpetuate reduced levels of native plant species richness, it does more closely resemble the structure and disturbance regimes of native communities compared to cheatgrass monocultures. Therefore, recent trends toward the use of seed mixtures containing native species might ameliorate the reduction of native plant species richness (Pellant and Monsen 1993) and further approximate species compositions and disturbance regimes of native communities. Currently, the BLM normally uses native species in seeding mixtures where conditions are such that a native species mixture has a reasonable chance of establishment and persistence. Where native species mixtures are not expected to become established or persist in some of the more drier cheatgrass infestations, mixtures with crested wheatgrass are still used to control cheatgrass.

A proactive technique to reduce the cheatgrass–wildfire cycle is to seed strips of fire resistant vegetation (greenstripping) at strategic locations, in order to slow or stop the spread of wildfires (Pellant 1990). Herbaceous plant species commonly used in greenstripping include introduced wheatgrasses, Russian wildrye, dryland alfalfa, lewis flax, and small burnet (Pellant 1994). Greenstripping is not a solution to the cheatgrass–wildfire cycle, but it can help reduce the size and frequency of wildfires. The ecological benefits of greenstripping include conservation of native plant species richness and shrub cover on fire-prone landscapes, and the eventual enhancement of native plant species richness (West 1979, Whisenant 1990, Young and Evans 1978).

Cheatgrass distribution and dominance continue to expand, particularly in the dry forest, dry shrub, and dry grass PVGs. Although cheatgrass tends to form a

Table 2-35. Susceptibility of Broad-scale Cover Types to Invasion by Weed Species.[1]

Cover Type	Brte[2]	Canu	Caspp	Cedi	Cema	Cere	Ceso	Cevi	Chju	Chle	Ciar	Civu	Crvu	Eues	Hagl	Hiau	Hipr	Isti	Lida	Livu	Lysa	Onac	Pore	Saae	Taas
Alpine Tundra	L[3]	L	L	L	L	L	L	L	L	L	M	M	L	L	L	M	L	L	M	L	L	L	L	U	L
Aspen	M	M	M	M	M	M	M	M	M	M	H	M	M	M	M	M	M	M	M	M	L	U	M	U	M
Big Sagebrush	H	U	M	M	M	M	M	M	M	U	M	M	L	M	M	L	L	H	M	M	L	M	U	H	M
Bitterbrush/ Bluebunch Wheatgrass	H	M	M	H	M	U	M	M	U	U	M	M	M	M	M	L	L	M	M	M	L	U	M	U	M
Chokecherry/ Serviceberry/Rose	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	H	M	M	M	M	M	U	M
Cottonwood/Willow	M	M	M	M	H	M	M	M	M	H	H	M	L	H	L	M	H	M	M	M	M	M	M	U	M
Cropland/ Hay/Pasture	M	M	H	M	H	M	H	M	H	M	M	M	M	H	M	M	M	H	M	M	L	M	M	U	M
Engelmann Spruce/ Subalpine Fir	H	H	M	M	M	M	M	M	M	H	H	H	L	M	L	M	M	L	M	M	M	U	M	U	M
Exotic Forbs/ Annual Grass	H	M	H	M	M	M	H	H	H	M	M	M	M	M	M	M	M	H	H	H	M	M	H	M	M
Fescue-Bunchgrass	H	H	M	H	H	M	H	M	M	M	H	H	M	H	M	L	L	H	H	H	L	M	H	H	M
Grand Fir/White Fir	M	M	M	M	M	M	M	M	M	M	M	M	L	M	L	M	U	L	M	M	M	U	M	U	M
Herbaceous Wetlands	M	M	M	M	H	M	H	M	L	H	H	M	L	M	L	H	M	M	M	H	H	M	H	U	M
Interior Douglas-fir	H	H	M	M	H	M	M	M	M	M	H	H	M	M	M	M	M	M	M	M	L	M	H	U	M
Interior Ponderosa Pine	H	M	M	H	H	M	M	M	M	M	M	M	M	M	M	L	L	M	M	M	L	M	H	U	M
Juniper/Sagebrush	M	M	M	M	M	U	M	M	M	U	M	M	L	M	M	L	L	H	M	M	L	U	U	U	M
Juniper Woodlands	M	M	M	M	M	U	M	M	M	U	M	M	L	M	M	L	L	M	M	M	L	U	M	U	M
Limber Pine	M	M	M	M	M	M	M	M	M	M	M	M	M	L	M	M	L	M	M	M	L	L	M	U	M
Lodgepole Pine	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	L	M	M	M	L	M	M	U	M
Low Sagebrush	M	U	M	M	U	U	M	M	U	U	M	M	L	M	M	L	L	H	M	U	L	U	U	U	M
Mixed-Conifer Woodlands	H	M	M	M	H	M	M	M	M	U	H	M	L	M	M	L	L	M	M	M	L	U	H	U	M
Mountain Big Sagebrush	H	M	M	M	M	M	M	M	M	U	M	M	L	M	M	L	L	H	M	M	L	M	M	U	M
Mountain Hemlock	M	M	M	M	M	M	M	M	M	M	M	M	L	L	L	M	M	L	M	M	M	L	M	U	M
Mountain Mahogany	M	M	M	M	M	M	H	M	U	U	M	M	M	M	M	L	L	M	H	M	L	U	M	H	M
Native Forb	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	L	L	M	M	M	M	M	H	U	M
Oregon White Oak	M	U	M	M	M	M	M	M	M	M	M	M	M	M	M	L	L	M	U	M	L	M	M	U	M

Table 2-35. Susceptibility of Broad-scale Cover Types to Invasion by Weed Species.[1] (continued)

Cover Type	Brte[2]	Canu	Caspp	Cedi	Cema	Cere	Ceso	Cevi	Chju	Chle	Ciar	Civu	Crvu	Eues	Hagl	Hiau	Hipr	Isti	Lida	Livu	Lysa	Onac	Pore	Saae	Taas
Pacific Ponderosa Pine	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	L	L	M	M	M	L	M	M	U	M
Pacific Silver Fir/ Mountain Hemlock	M	M	M	M	M	M	M	M	M	M	M	M	L	L	L	M	M	L	M	M	M	L	M	U	M
Red Fir	M	M	M	M	M	M	M	M	M	M	M	M	L	L	L	M	M	L	M	M	M	L	M	U	M
Salt Desert Shrub	M	M	M	L	L	M	L	M	L	L	M	M	L	M	H	L	L	L	L	L	L	L	L	U	L
Shrub or Herb/ Tree Regen	M	M	M	M	M	M	M	M	H	M	M	M	M	M	M	M	L	M	M	M	L	M	H	U	M
Shrub Wetlands	M	H	M	M	H	M	M	M	L	M	H	H	L	M	L	M	M	M	M	M	H	M	M	U	M
Sierra Nevada Mixed-Conifer	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	L	L	M	M	M	L	M	M	U	M
Western Larch	M	M	M	M	M	M	M	M	M	M	H	M	M	M	M	M	M	M	M	M	L	M	M	U	M
Western Redcedar/ Western Hemlock	M	H	M	M	M	M	M	M	M	H	H	H	L	M	L	M	M	L	M	M	M	U	M	U	M
Western White Pine	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	L	U	M	U	M
Wheatgrass Bunchgrass	H	M	M	H	H	M	H	M	M	M	H	M	M	M	M	L	L	H	H	M	L	M	H	H	M
Whitebark Pine	L	L	L	L	L	L	L	M	L	L	M	M	L	L	L	M	L	L	M	L	L	L	L	U	M
Whitebark Pine/ Alpine Larch	L	L	L	L	L	L	L	L	L	L	M	M	L	L	L	M	L	L	M	L	L	L	L	U	L

[1] Broad-scale Cover Types in the basin and their susceptibility to invasion by 25 weed species (24 legally declared noxious, plus cheatgrass).

[2] Species codes for exotic plants: Brte = cheatgrass; Canu = musk thistle; Caspp = whitetop; Cedi = diffuse knapweed; Cema = spotted knapweed; Cere = Russian knapweed; Ceso = yellow starthistle; Cevi = squarrose knapweed; Chju = rush skeletonweed; Chle = oxeye daisy; Ciar = Canada thistle; Civu = bull thistle; Crvu = common crupina; Eues = leafy spurge; Hagl = halogeton; Hiau = orange hawkweed; Hipr = yellow hawkweed; Isti = Dyers woad; Lida = dalmatian toadflax; Livu = yellow toadflax; Lysa = purple loosestrife; Onac = Scotch thistle; Pore = sulfur cinquefoil; Saae = Mediterranean sage; Taas = medusahead.

[3] Ratings representing susceptibility to invasion, and definitions:

H = High susceptibility to invasion — Exotic plant species is an “invader” and invades the cover type successfully and becomes dominant or codominant even in the absence of intense or frequent disturbance;

M = Moderate susceptibility to invasion — Exotic plant species is a “colonizer” and invades the cover type successfully because high intensity or frequency of disturbance impacts the soil surface or removes the normal canopy cover;

L = Low susceptibility to invasion — Exotic plant species typically does not establish because the cover type does not provide suitable habitat; and

U = Unknown susceptibility to invasion — Herbarium mount labels did not report the species at the collection site that existed in association with the mounted exotic plants, or ecological requirements of the exotic plant are not available in the literature, or there was a lack of distribution records (for example, herbaria mounts) for the exotic plant, or the extent of the cover type in the Project Area might be so minor as to prevent or restrict the probability of obtaining distribution records for the exotic plant within that cover type.

Source: Hann, Jones, Karl, et al. (1997).

Table 2-36. Susceptible Cover Types Description[1].

Cover Type	Description
Alpine Tundra	*Phyllodoce* spp. (low shrubs)
Aspen	*Populus tremuloides*
Barren	Rock/Barrenlands
Big Sagebrush	*Artemisia tridentata wyomingensis* *Artemisia tridentata tridentata/Elymus cinereus* *Artemisia tripartita/Agropyron cristatum* *Artemisia tripartita/*Exotic Herbs *Artemisia tridentata tridentata/Agropyron* spp. *Artemisia tridentata tridentata/Bromus tectorum* *Artemisia* spp./*Bromus tectorum* *Artemisia tripartita*
Bitterbrush/Bluebunch Wheatgrass	*Purshia tridentata/Bromus tectorum* *Purshia tridentata/Agropyron spicatum*
Chokecherry/Serviceberry/Rose	*Prunus virginiana/Amelanchier alnifolia/Rosa* spp.
Cottonwood/Willow	*Populus trichocarpa*/*Salix* spp. *Populus* spp./*Cornus* spp. *Populus* spp./*Poa pratensis*
Cropland/Hay/Pasture	Dryland Crop Dryland Pasture/Hayland Irrigated Crop Irrigated Pasture/Hayland
Engelmann Spruce/Subalpine Fir	*Picea engelmannii/Abies lasiocarpa*
Exotic Forbs/Anual Grass	Exotic Forbs Exotic Grass (*Bromus tectorum/Taeniatherum caput-medusae/Poa secunda*) Exotic Herbaceous Exotic Herbs Exotic Perennial Grass
Fescue-Bunchgrass	*Festuca idahoensis*/*Agropyron* spp. Low Productivity Perennial Grass Perennial Native Bunchgrass Perennial Native Herbaceous Seeded Native Grass (*Agropyron spicatum/Festuca idahoensis*) Seeded Native Grass (*Poa secunda/Agropyron spicatum*) Small Perennial Grass
Grand Fir/White Fir	*Abies grandis/Abies concolor*
Herbaceous Wetlands	*Carex nebraskensis* *Carex rostrata/Carex aquatilis* Grass/*Carex* spp. *Elymus* spp.
Interior Douglas-fir	*Pseudotsuga menziesii* var. *glauca* *Pseudotsuga menziesii/Abies grandis*/Exotic Herbs *Pseudotsuga menziesii/Abies grandis*/*Populus* spp./Shrub
Interior Ponderosa Pine	*Pinus ponderosa* var. *scopulorum* *Pinus* spp./*Populus* spp./Exotic Herbs *Pinus* spp./*Populus* spp./Shrub
Juniper/Sagebrush	*Juniperus* spp./*Artemisia arbuscula*/*Festuca idahoensis*/Forb *Juniperus* spp./*Artemisia* spp./*Agropyron* spp.
Juniper Woodlands	*Juniperus* spp./Exotic Herbs *Juniperus* spp./*Artemisia arbuscula*/Shortgrass *Juniperus* spp. Forest/Exotic Herbs *Juniperus* spp. Woodlands *Juniperus* spp./Native Bunchgrass *Juniperus* spp./*Poa secunda*
Limber Pine	*Pinus flexilis*
Lodgepole Pine	*Pinus contorta*
Low Sagebrush	*Artemisia arbuscula*/Native Forbs *Artemisia arbuscula*/*Bromus tectorum* *Artemisia arbuscula*/Native Bunchgrass *Artemisia* spp./*Poa secunda*
Mixed-Conifer Woodlands	Conifer/Exotic Herbs Conifer Encroachment/Exotic Grass Conifer Encroachment/*Artemisia* spp./Perennial Grass Conifer/Perennial Grass
Mountain Big Sagebrush	*Artemisia tridentata vaseyana*/Perennial Grass

Table 2-36. Susceptible Cover Types Description[1]. (continued)

Cover Type	Description
	Artemisia tridentata vaseyana/Exotic Herbs *Artemisia tridentata vaseyana*/Perennial Herbs
Mountain Hemlock	*Tsuga mertensiana*
Mountain Mahogany	*Cercocarpus* spp.
Native Forb	*Deschampsia* spp./*Calamagrostis* spp. Exotic Moist Herbs Exotic Riparian Herbs Native Forbs Pioneer Forbs
Oregon White Oak	*Quercus alba*/Exotic Herbs *Quercus alba*/Shrub
Pacific Ponderosa Pine	*Pinus ponderosa* var. *ponderosa*
Pacific Silver Fir/Mountain Hemlock	*Abies amabilis*/*Tsuga mertensiana*
Red Fir	*Abies magnifica* var. *shastensis*
Salt Desert Shrub	*Sarcobatusvermiculatus* *Sarcobatus vermiculatus/Distichlis stricta* Salt Desert Shrub[2]
Shrub or Herb/Tree Regen	General Shrub Grass/Forb Mid Shrub West Cascades Mountain Shrub - No other Mountain Shrub/*Ceanothus* spp. Shrub/Regen
Shrub Wetlands	*Cornus* spp./*Crataegus* spp. Gravel Bar *Salix* spp. low/*Carex* spp. *Salix* spp. low/Grass *Salix* spp./*Calamagrostis* spp. *Salix* spp./*Carex* spp./*Castor canadensis* *Salix* spp./*Poa pratensis* *Sarcobatus vermiculatus*
Sierra Nevada Mixed-Conifer	Sierra Nevada Mixed-Conifer
Urban	Urban Land
Water	Water
Western Larch	*Larix occidentalis*
Western Redcedar/Western Hemlock	*Thuja plicata /Tsuga heterophylla*
Western White Pine	*Pinus monticola*
Wheatgrass Bunchgrass	*Agropyron cristatum* *Agropyron cristatum*/*Bromus tectorum* *Agropyron spicatum* *Agropyron* spp./*Poa secunda* *Aristida longiseta* *Bromus tectorum* *Elymus cinereus* *Elymus cinereus*/*Agropyron* *Elymus cinereus*/*Bromus tectorum* Exotic Annual Grass Fire Maintained Grass (*Poa secunda/Agropyron spicatum*) Native Perennial Grass Perennial Herbs *Poa secunda/Festuca octoflora* *Poa pratensis* *Poa secunda* *Poa secunda*/Perennial Forbs Seeded Exotic *Agropyron* spp. *Sitanion hystrix*
Whitebark Pine/Alpine Larch	*Pinus albicaulis/Larix lyallii* *Pinus albicaulis/ Larix lyallii/Abies lasiocarpa*
Whitebark Pine	*Pinus albicaulis*

[1] Description of broad-scale cover types in the Basin used in Table 2-35 to characterize the susceptibility of vegetation types to invasion by weed species.
[2] Four representative plants in the Salt Desert Shrub type found within the Basin are *Eurotia lanata* (winterfat), *Atriplex confertifolia* (shadscale), *Elymus cinereus* (Great Basin wildrye), and *Grayia spinosa* (spiny hopsage).

Source: Hann, Jones, Karl, et al. (1997).

stable vegetation state after establishment, because of frequent fire, other exotic plants are invading cheatgrass-dominated communities and potentially degrading rangeland health even further. Examples of these other exotic plants include medusahead, yellow starthistle, and ventenata.

Composite Landscape Conditions

The condition of the interior Columbia Basin has changed in the last century. Changes in vegetation composition and structure, brought about by changes in succession / disturbance regimes, have led to dwindling populations of some aquatic and terrestrial species, substantial increases in others, and reduced capacity to achieve social and economic values. The project area is experiencing effects from disturbances that are not characteristic of the past when basin ecosystems were more balanced. These effects can be measured in a variety of different ecosystem variables and characteristics. Together, the effects can be interpreted to show a decline in overall landscape health.

One of the measures of landscape conditions is the departure of Historical Range of Variability (HRV departure). HRV departure is a comparison of how current patches in a subwatershed landscape differ from the normal range and variability of historical landscape patches in their vegetation composition and structure and succession / disturbance regime.

A set of broad-scale representative variables were used to assess the "composite HRV departure" of vegetation patches. These variables include vegetation composition, structure, size, contagion (proximity to other patches of vegetation), succession and disturbance processes, in the context of whether the biophysical setting is appropriate, based on the findings of Hessburg et al. (1999) and Hann et al. (1997). The intent was to integrate vegetation patch size, shape, composition, structure, environment, fragmentation, contagion, and succession / disturbance regime into one index for each subwatershed (Hemstrom et al. 1999). These individual subwatershed indexes were then added to achieve an estimate of the broad-scale composite HRV departure.

In general, BLM- and Forest Service-administered lands are less departed from historical conditions than other lands due mainly to agriculture and other development on much of the lands not administered by the BLM or Forest Service. Over half the BLM- and Forest Service-administered lands in the project area are currently in a high or moderate HRV departure class (see Figure 2-24), which means they are moderately or highly different than historical conditions. Large areas of high departure on BLM- and Forest Service-administered lands can be found in the Butte RAC, Eastern Washington RAC, Upper Columbia-Salmon Clearwater R1 RAC, and the Klamath PAC (see Map 2-35).

The ecological integrity trend variable (Quigley et al. 1999) used in the Supplemental Draft EIS is generally equivalent to the ecological integrity variable (Quigley et al. 1996 and 1997) as defined in the Scientific Assessment and Draft EISs. It is based on the average trends of subwatershed composite HRV departure, aquatic habitat conditions, and road density. Using this measure, half of the BLM- and Forest Service-administered lands in the project area are currently classified as having low ecological integrity (see Figure 2-25). The highest concentration of subwatersheds in the high ecological integrity category can be found in the Eastern Washington-Cascades PAC and Upper Columbia-Salmon Clearwater-R4 RAC (see Map 2-36).

Landscape health is defined by Hann et al. (1999) as "the best fit of the dynamic interaction of human land use, biodiversity, and ecosystem health that is in balance with the limitations of the biophysical system and inherent disturbance processes." In this analysis all the subwatersheds in the project area currently fall into the moderate, low, and very low landscape health categories; none are high or very high (see Figure 2-26). The highest concentration of subwatersheds in the moderate category of landscape health can be found in the Eastern Washington-Cascades PAC and Upper Columbia-Salmon Clearwater-R4 RAC (Map 2-37).

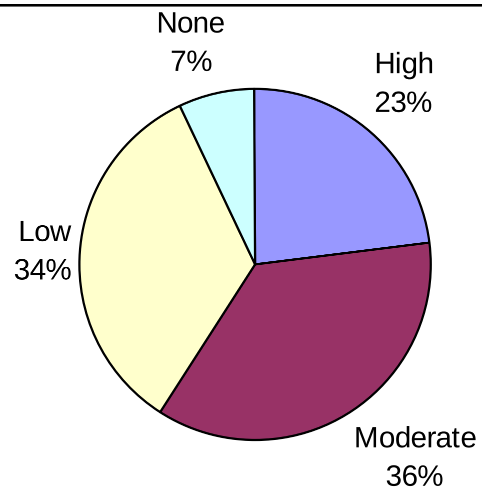

Figure 2-24. Current HRV Departure Class, Forest Service- and BLM-administered Lands.

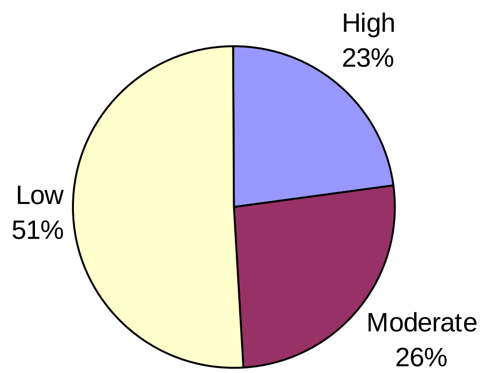

Figure 2-25. Current Ecological Integrity, Forest Service- and BLM-administered Lands.

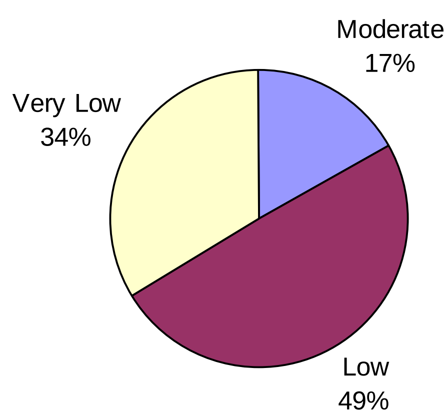

Figure 2-26. Current Landscape Health Conditions, Forest Service- and BLM-administered Lands.

Map 2-35. Historical Range of Variability Departure Classes: Current.

Map 2-36. Ecological Integrity: Current.

Map 2-37. Landscape Health: Current.

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